A review of options for the containment, control and eradication of illegally introduced smallmouth bass (*Micropterus dolomieu*)

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ABSTRACT

Halfyard, E.A. 2010. A review of options for the containment, control and eradication of illegally introduced smallmouth bass (*Micropterus dolomieu*). Can. Tech. Rep. Fish. Aquat. Sci. 2865: vi + 71 p.

This report is a review of literature regarding options for the containment, control and eradication of non-native fish species. Containment options consist of physical and behavioural barriers. Physical barriers offer the highest efficiency but may be labour intensive. Behavioural barriers may be relatively labour-free but they are prone to failure. Control is defined as the removal of some portion of the population, usually to reduce abundance or biomass. Eradication is defined as the local extirpation of a population within a confined or designated area. Control options include mechanical removal, biological removal, environmental removal, and chemical piscicides. Mechanical removal is often easily applied though laborious and only marginally effective. Biological removal is largely untested and generally risky. Environmental removal via water removal is highly effective but often unfeasible. Chemical piscicides are highly effective but they may impact non-target aquatic species. Piscicides and complete dewatering offer the greatest potential for eradication. The deployment of multiple techniques may increase the probability of successful management of non-native species. The probability of successful containment, control or eradication of undesirable fishes decreases with time post-arrival and early intervention is preferable.

RÉSUMÉ

Ce document est une revue de la littérature des options pour contenir, contrôler et extirper des poissons non indigènes. Les options pour contenir les espèces non indigènes consistent de barrières physiques et de barrières qui modifient le comportement. Les barrières physiques ont une plus grande efficacité mais peuvent nécessiter des travaux intensifs. Les barrières qui modifient le comportement peuvent nécessiter moins de travaux intensifs mais sont susceptibles à faillir. Le contrôle est l'élimination entière d'une population dans un endroit limité ou désigné. Les options pour le contrôle comprend des méthodes d'enlèvement de nature mécanique, biologique, environnemental et chimique (poisons). Les méthodes mécaniques sont simples à mettre en pratique mais sont très souvent laborieuses et peu efficaces. Les méthodes biologiques n'ont pas fait leurs preuves et sont risquées. Les méthodes environnementales, qui consistent à retirer l'eau, sont très efficaces mais souvent infaisables. Les poisons sont très efficaces mais ont aussi un impact sur les espèces non ciblées. L'utilisation de poison avec les méthodes environnementales ont le plus grand potentiel d'extirpation. L'utilisation de multiples techniques offre une plus grande probabilité de gérer avec succès les espèces non indigènes. L'intervention précoce est préféré car la probabilité de contenir, contrôler ou éradiquer les espèces non indigènes diminue avec le temps après l'arrivée.

PREFACE

This report is the product of a contract for services issued by Fisheries and Oceans Canada (DFO) under the scientific authority of Gérald Chaput (DFO). The report was prepared in fulfillment of one of the terms of reference for a risk assessment of the introduction of smallmouth bass in a headwater lake in a renowned Atlantic salmon river in New Brunswick. A draft of this report was presented at the science peer review meeting of the Regional Advisory Process in Moncton (NB), January 27-28, 2009. The objectives of the meeting were: (1) to review the historical and current distribution of smallmouth bass, (2) its biology, habitat requirements and availability of habitat to smallmouth bass in the Maritime provinces, (3) to examine evidence for negative interaction between Atlantic salmon and smallmouth bass, (4) to conduct a risk assessment of smallmouth bass impacts on the ecosystem of the rivers of the Gulf Region, with a specific risk analysis of impacts to Atlantic salmon in the Southwest Miramichi River, and (5) to review options for and the effectiveness of mitigation measures for minimizing the risks associated with range extension of smallmouth bass. The risk assessment followed the guidelines established by the DFO Centre of Expertise for Aquatic Risk Assessment (CEARA) for assessing the biological risk of aquatic invasive species in Canada. The advice from the meeting was published as a science advisory report of the DFO Canadian Science Advisory Secretariat (http://www.dfo-mpo.gc.ca/csas-sccs/).

The opinions and interpretation of the data and information expressed in this report are those of the author and do not necessarily reflect the views of the Fisheries and Oceans Canada. The use of trade names or product does not constitute endorsement by the author or Fisheries and Oceans Canada.

Species identification in this paper follows the integrated taxonomic information system (ITIS) for common and scientific names and the list of common names used in this paper and their associated scientific names is provided as an appendix.

1.0 INTRODUCTION

The Miramichi River, New Brunswick, is the most important recreational fishing area in New Brunswick and the Maritime provinces. Of the eight recreational fishing areas in the province, the Miramichi area receives the most angling effort, estimated at 135,000 resident plus non-resident angler days in 2000, the majority directed at Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) (Cronin et al. 2001). Many private angling camps operate in the area and numerous commercial outfitters operate on the Miramichi River. Angling within the Miramichi recreational fishing area in 2000 generated an estimated CAN $20 million in annual fishing-related expenditures, of which over 30% was attributed to non-resident anglers (Cronin et al. 2001).

Non-native fishes often out-compete native fishes through a variety of mechanisms such as direct predation (Rieman et al. 1991, Ogutu-Ohwayo 1993, Fritts and Pearson 2006), inter-specific competition (Knapp 1996, Bohn and Amundsen 2001, Bigelow et al. 2003), hybridization (Simberloff 1996, Perry et al. 2002, Konishi et al. 2003), introduction of disease and pathogens (Gozlan et al. 2005, NYDEC 2008), and trophic regime shifts (Roemer et al. 2001).

On September 26th 2008, an angler captured a 20 cm smallmouth bass (*Micropterus dolomieu*) in Miramichi Lake and reported it to the New Brunswick Department of Natural Resources (NBDNR). Miramichi Lake (46.46° N, 66.97° W) is a 221 hectare headwater lake of the Southwest Miramichi (Figure 1). This was the first report of smallmouth bass in the Miramichi system. Smallmouth bass introductions in other parts of Canada are considered to pose a substantial threat to the native aquatic communities (Brown et al. 2009, Tovey et al. 2008, McNeill 1995).

To confirm the presence of smallmouth bass in the lake, NBDNR conducted a survey of the lake from September 29th to October 3rd 2008. Five young-of-year Smallmouth Bass were captured via electrofishing and another 2 smallmouth bass of age unknown were captured in the gill nets. To assess the extent of dispersal from the lake into other parts of the watershed, a stream survey was conducted from October 6th to 10th 2008. Using backpack electrofishing equipment, selected sites on the Southwest Miramichi and the 5.3 km stream that connects Miramichi Lake to the Southwest Miramichi, named Lake Brook, were sampled. Young-of-year, reported to be in "good condition", were discovered though their distribution appeared to be limited to the first 300m downstream from Miramichi Lake. The stream in this section is slow flowing, deep and bounded on the downstream end by riffle habitat. From that point downstream, habitat is generalized as typical riffle-pool sequence.

Based on surveys conducted in September and October, and initial confinement procedures, it appeared that smallmouth bass in Miramichi Lake were currently in low abundance and their distribution outside of the lake may be limited.

In a temporary effort to limit the spread of smallmouth bass from Miramichi Lake, the watershed group (Miramichi Watershed Management Committee) with partner support erected a portable conduit counting fence fitted with 1.9cm mesh on Lake Brook. This trap operated from October 23rd to November 27th 2008. Because this illegal introduction was detected early, management programs to control the spread and/or establishment of smallmouth bass may prove successful.

This document presents a literature review of available containment, control and eradication options. The timeframe allotted for this review was compressed and as a result, the scope of this

review is as complete as possible within the timeframe allotted. Although this review was catalyzed by the introduction of smallmouth bass to the Miramichi system, technologies discussed in sections 2 and 3 are widely applicable and have been considered elsewhere for managing other species of undesirable, non-native fish.

1.1 Impetus of Review

A fundamental problem facing fisheries managers when investigating management options for non-native fish introductions is that much of the literature exists outside peer-reviewed primary literature. Furthermore, much of the technological information specific to management options is reported by engineering or management professionals focused on industrial applications (i.e. reducing fish entrainment at hydroelectric plants) and biological implications are seldom considered. Finally, many reports fail to monitor long term impacts non-native management efforts and management efficacy is rarely evaluated. Meronek et al. (1996) stated similar frustrations in their review of 250 fish control projects, where 25% of the reviewed projects provided insufficient evidence of post-treatment status of the target fish (es).

The future of fisheries management will be faced with large-scale environmental concerns such as global climate change, human population expansion and a continued global homogenization of aquatic biodiversity. The management of non-native fishes will undoubtedly remain an important mandate for managers across jurisdictions. As such, available information regarding management options must become widely available and, more importantly, future management programs must be adequately assessed to refine the containment, control and eradication of undesirable non-native fishes.

1.2 Managing Non-Native Fishes

The intentional introduction of fish has in the past been intended to increase food supplies, create or enhance fishing opportunities, as biological control, and for aesthetic purposes (Li and Moyle 1999). Fish have also been accidentally introduced through the release of bait, ballast water and aquarium fishes (Lennon 1971, Brock and Crisman 1991).

Non-native fish introductions, even intentional ones, often create unexpected problems. An early example is the introduction of common carp to lakes in the San Francisco area of California (Smith 1896 *in* Cole 1904). Smith reported that these lakes, previously used as water supplies, had become so turbid as a result of the carp that they were of little use and had lost their aesthetics. As a result, the first program to control non-native fishes was initiated in Merced Lake, California in 1891 (Smith 1896 *in* Cole 1904). Control efforts of the day included seine netting and the introduction of "19 good sized seals", presumably sea lions. Follow-up seining in 1895 indicated that no carp were present and the author considered the control efforts successful.

Some 18 years later, Titcomb (1914) would describe the first application of chemical piscicides (copper sulfate) to control "obnoxious" fishes in a Vermont lake. The use of piscicides thereafter gained widespread popularity among fisheries managers and it was commonly deployed for the control of competitors to game fish (i.e. Smith 1940, Hayes and Livingstone 1955).

Efforts to control undesirable fish (presumably some of which were non-native) using piscicides as well as other methods, such as mechanical removal, biological removal, and electrofishing, also gained popularity during the first half of the 20th century (Wydoski and Wiley 1999).

These early efforts were primarily aimed at increasing the quality of angling for "game fish" species or improving catches in commercial inland fisheries (Wydoski and Wiley 1999). Since then however, efforts to develop and apply effective management measures to mitigate the effects of non-native fishes have grown concurrently with non-native fish expansion. Presently, the management of non-native fishes is one of the most widely distributed fisheries management related activities, with active programs worldwide.

The choice of mitigation measures is generally determined in consideration of numerous factors such as the risk to native fishes, the potential to achieve management objectives, public opinion, financial restraints and potential impacts on native non-fish organisms. The management of introduced, un-desirable fishes can be grouped into three categories: containment, control, and eradication.

Containment is generally the primary response in non-native fish management and is often intended to prevent or delay further spread of non-native species to connected habitats. Containment does not address the presence of the non-native fish in affected habitats and is achieved by physically restricting fish to a specified area or by creating conditions that discourage migration.

Control of non-native fishes involves the removal of individuals at less than 100% efficiency. Control is often aimed at reducing the negative impacts of non-native fish on the surrounding ecosystem. Proxies for success in control efforts may include: reductions in standing stock of target species; reductions in proportional stock density (PSD); shifts in relative weight or condition of target species; shifts in relative weight or condition of native fishes; shifts in size structure of non-native or native fishes; increased survival of native fishes or some other project-specific objectives.

Eradication of non-native fish involves the complete remove of target species within a specific area. This is often warranted where containment is unlikely, where the potential for establishment is high, where the risk of non-natives spreading to neighbouring habitats is substantial and where the potential ecological risks of the non-native species is unacceptably high.

2.0 CONTAINMENT OF NON-NATIVE FISHES

2.1 Introduction to Barriers

Where control or eradication of non-native fishes is logistically difficult, or as the first response, the segregation of native fish and non-native fish (colonized & un-colonized habitats) may provide the most feasible management strategy to address impacts on native species.

Currently, artificial and altered natural barriers are common conservation tools for protecting native fish habitat from invasion by non-native fishes. Dispersal of non-native fish dispersal may occur in either downstream or upstream directions. Most reports highlight upstream migration (e.g. Kruse et al. 2001, Thompson and Rahel 1998) although a few describe dispersal from upstream introductions (Adams et al. 2001). This is because many intentional and unintentional introductions primarily occur in the lower portions of watersheds. For example, industrial shipping often utilizes ports at the mouths of large river systems (i.e. Delaware River, Hudson River, Sacramento River, Fraser River). Non-native species released from ballast water in ships then colonize in an upstream direction. The round goby is a good example of this and its dispersal occurred in a dendritic fashion originating in the St. Clair River (Marsden and Jude 1995, Raloff 1995).

Accordingly, many barrier technologies focus on preventing upstream movements of fish. Less demand for downstream-movement barriers, in addition to several technical difficulties that make preventing downstream movement of fish more difficult than upstream movement, have slowed their development. The technical difficulties are due to the nature of flowing water itself, aiding downstream movement and impeding upstream movement. In an assessment of instream movements of non-native brook trout, Adams et al. (2001) found that brook trout dispersed downstream across gradients of up to 80% and down an 18m waterfall. Conversely, Kondratieff and Myrick (2006) found that brook trout (moving in an upstream direction) could only traverse vertical waterfalls of 0.73 m in height. Active migration is a necessity for upstream migration, but not necessarily for downstream movement, as has been theorized by Good and Cargnelli (2004).

Flooding poses unique challenges for both upstream and downstream dispersal barriers. High water can aid movement by providing alternate routes around barriers to upstream movement, and also by washing fish downstream. There are examples of alternate upstream barrier structures to compensate for side channel formation during high water events (i.e. Riley et al. 1999, 2000, 2004); however the prevention of downstream dispersal during floods has not been adequately addressed.

Industrial applications of fish barriers may provide some insight to potentially limiting downstream movements of fish. In particular, hydroelectric developments have invested substantial time and resources into reducing fish entrainment (intake into turbines). However, few researchers have explored options for large-scale dams as directed downstream fish passage barriers, though there is evidence to suggest that they may be ineffective under most operating conditions (Brunson et al. 2007).

Technologies to permit downstream fish passage (or conversely containment) are much less advanced than for upstream fish passage (containment) (Larinier and Travade 2002), and due to the difficulties preventing downstream migration, barrier projects have focused primarily on preventing upstream dispersal of non-native fishes. For example, authorities in the Southwest

United States have installed at least 75 lowhead dams and artificial waterfalls in an effort to prevent the spread of introduced non-native fishes, namely smallmouth bass, green sunfish and bullheads (*Ameiurus* spp.) among others (Carpenter and Terrell 2005). Similarly, the National Park Service (NPS) of the United States Department of the Interior actively installs upstream barriers to protect native salmonid species or uses existing upstream barriers as integral parts of their conservation programs. Often, the goal is to create upstream refuges where native salmonids could be segregated from the invading non-native fishes (R. Clarkson, US Bureau of Reclamation, pers. comm.). Rocky Mountain National Park is the site of a reclamation initiative to restore habitat for the native greenback cutthroat trout and Colorado River cutthroat trout and to reduce inter-specific competition with introduced brook trout. The NPS remodelled a natural barrier to provide a greater barrier to upstream moving fish (Myrick and Kondratieff 2003). Similarly, non-native salmonid species were removed above barriers constructed to prevent upstream movements in Great Basin National Park (Baker et al. 2008) and in Great Smoky Mountain National Park (Moore et al. 1986).

Large-scale dams inherently limit the upstream movement of fishes and managers exert significant effort to reduce the effects of habitat segregation in native, desirable fishes, particularly anadromous fishes such as salmon. Without the use of fish ladders or manual transportation, the upstream movements of undesirable fishes would presumably be blocked by large dams.

Barriers to upstream movement have also been used to create refugia in headwater sections of their native fish habitats as a conservation measure against invasive non-native fishes (i.e. Kruse et al. 2001, Novinger and Rahel 2003, Thompson and Rahel 1998, *among others*). This technique permits downstream movement of native fishes but no upstream movement of any fish. There is some debate however that the isolation of native fishes from invading non-native fishes using barriers may not offer ideal or long-term solutions. In an assessment of headwater isolation strategies for Yellowstone cutthroat trout in Wyoming, Kruse et al (2001) concluded that the effective population size in isolated areas is often insufficient to reduce the risk of extinction as a result of genetic limitations. Furthermore, of the isolation-strategy restoration projects they reviewed, they postulated that efforts appeared to have a low probability of long term preservation of target trout populations. The long-term use such upstream barriers may result in unidirectional gene flow (Shaw et al. 1994).

In systems where diadromous fishes are present, additional considerations regarding fish passage are warranted as barriers may limit their upstream distribution upon return from the ocean. For species such as Atlantic salmon, pacific salmon (*Oncorhynchus* spp.) and several members of the herring family (*Alsoa* spp.), system-wide juvenile production may be reduced as a result of lost spawning habitat. Conversely, for American eels, adult habitat may be lost as returning juvenile eels (elvers) may be restricted from these upstream sections.

The second major division of barrier technologies, after the direction of movement, is the dichotomy between physical diversion and behavioural diversion. Physical diversion contains fish by providing a tangible barrier to their movement. Conversely, behavioural barriers are porous in nature and the deterrence of fish is accomplished by creating conditions that are unfavourable or discouraging, thus repelling fish from the area. This may be accomplished by emitting signals or cues or by manipulating hydraulic conditions so that fish could theoretically pass if "intent", but are discouraged against attempting to pass. Behavioural barriers have

received increased attention (Sager et al. 2000b) as they are often economical and perceived as preferential (Turnpenny et al. 1998).

Managers faced with restricting the movement of undesirable fishes have several barrier options. The two most common physical options are hydraulic barriers or screen, net and weir barriers. If non-physical barriers are preferred, electrical barriers, bubble barriers, acoustic barriers and light-based technologies have been explored. Their use has been restricted primarily to experimental installations. A handful of other technologies have been discussed with little field evaluation.

2.2 Physical Barriers

2.2.1 Hydraulic barriers

Hydraulic barriers exclude fish by exceeding some physiological threshold of the target fish, usually leaping ability. For example, barrier water falls are vertical drops in water level that exceed the jumping capabilities for upstream moving fish. Velocity barriers are constriction points that provide only one route around a barrier which is at an accelerated velocity above the maximum swimming speed of the target fish over some distance (Great Lakes Fishery Commission 2009). These barriers are often directly constructed (as purposeful barriers), indirectly constructed (i.e. poorly designed bridges), and also occur naturally. Natural barriers including water falls, high gradient rapids and other hydraulic barriers limit the natural distribution of fish species (Kruse et al 1997). Thus strategically placed hydraulic barriers may be similarly used to limit the distribution of invasive species (Hoover 2008).

Large scale hydraulic barriers are not frequently constructed for the sole purpose of restricting upstream dispersal of undesirable fishes. However, hydroelectric dams across much of the world inadvertently act as fish migration barriers (Gehrke et al. 2002, Morita and Yamamoto 2002) and there is substantial concern of their effects on native fishes (Bullen and Carlson 2003). The examination of large hydroelectric dams as downstream barriers has primarily focused on reducing entrainment of fish into hydroelectric turbines (Bullen and Carlson 2003).

Based on experimental measurements of leaping capabilities and to force hydraulic obstacles (swimming speeds), species-specific barriers obstructing migration may be constructed (Holthe et al. 2005). In a Norwegian experiment on the leaping capabilities of the invasive European minnow and the native brown trout, Holthe et al. (2005) suggested that waterfall barriers could be constructed to restrict passage for the small cyprinid yet permit passage ofbrown trout during their respective spawning migrations.

Scientists from the US Army Corps of Engineers evaluated the holding-station capabilities of the invasive round goby and its barrier traversing capabilities. Their results suggest that hydraulic barriers exceeding biologically-derived physiological thresholds would prevent upstream dispersal of gobies (Hoover et al. 2003).

The leaping capabilities of non-native brook trout were assessed in a laboratory setting, modeled and subsequently field-verified using an artificial barrier in Rocky Mountain National Park (Myrick and Kondratieff 2005). Their theoretical model underestimated the rate of passage of brook trout. Furthermore, contrary to the expected outcome based on laboratory tests, smaller trout (<20 cm) traversed the barrier at a higher rate than did large trout (>20 cm). This was

thought to be a function of the success of alternative passage strategies of the smaller trout (i.e. interstitial substrate migration vs. leaping).

Spurred by laboratory and field results or general conceptual designs, many hydraulic barriers have been constructed to prevent the upstream dispersion of undesirable fish species including non-native smallmouth bass (Boucher 2006, Josephson et al 2001, R.E. Clarkson, US Bureau of Reclamation, Arizona, pers. comm., Dec. 2008), non-native brook trout (Myrick and Kondratieff 2005, Thompson and Rahel 1998), non-native brown trout (Pister 1991), non-native sucker (Catostomus spp.), numerous cyprinidae (Baxter et al. 2003), green sunfish (Riley and Clarkson 2006), sea lamprey (Porto et al. 1999, Swink 1999), among others.

A Bonneville cutthroat trout restoration project in Great Basin National Park used existing human-made and natural barriers as divisions for treatment areas in which they successfully removed invasive brook trout, brown trout and rainbow trout (Baker et al. 2008). Projects in Great Smoky Mountain National Park used natural barriers to delineate stream reaches in which they mechanically and chemically removed invasive rainbow trout (*Oncorhynchus mykiss*) to reclaim important native brook trout habitat (Moore et al. 1986).

Assessments of hydraulic barriers have shown that when well constructed and maintained, they are highly effective. For example, Thompson and Rahel (1998) assessed the effectiveness of four gabion-type barriers in restricting upstream migration of non-native brook trout. They concluded that three of the four barriers were 100% efficient and the remaining ineffective barrier was considered poorly constructed. Poorly designed barriers were also identified as causes of barrier failures in a report produced by Young et al. (1996) when they found non-native fish above 20 out of 89 Wyoming barriers constructed to prevent upstream movement. Presumably, the remaining 69 barriers either contained non-natives in only the downstream area or non-native fishes did not attempt to cross those barriers.

In an assessment of the barriers constructed in Apache-Sitgreaves National Forest, Robinson et al. (2004) indicated that 64% of the 13 barriers they assessed failed to exclude non-native fish from upstream sections. However, most of the barriers that failed were reported to be in need of repair or reconstruction. Long-term maintenance costs presumably limited their success.

The ability of seven species of "rough" fishes (non-sportfish) to traverse a 2 m hydraulic barrier located on the Roaring River, Tennessee was examined by Bulow et al. (1988). Over two years, 1,056 fish were tagged below the barrier and 748 fish were examined above the barrier, of which none were marked. The authors concluded that the barrier was 100% effective at preventing the upstream movement of undesirable fishes.

Hydraulic barriers are most commonly constructed in small 1^{st} to 3^{rd} order streams (i.e. Carpenter and Terrell 2005, Moore et al. 1986, Myrick and Kondratieff 2003, Thompson and Rahel 1998) or mid-sized 4^{th} -5^{th} order streams (i.e. Baxter et al. 2003, Bulow et al. 1988). Streams of this size offer two logistic advantages over larger rivers. First, barriers in small streams tend to be smaller in scale and thus are easier to design, fund and construct than larger initiatives. Secondly, small streams tend to have higher gradient than higher order streams, facilitating the construction of waterfalls or spillways. At lower gradients, the upstream pooling associated with only moderate damming can transform riffle and run stream sections into pool habitat or, in more extreme circumstances, create impoundments that may inundate extensive areas of riparian habitat.

In summary, hydraulic barriers offer a high potential for containment of non-native fishes when the objective is to prevent upstream migration. This appears to be commonly accepted as hydraulic barriers are extensively used in the South-western and Western United States and commonly used elsewhere. However, for the control of downstream migrating fishes, hydraulic barriers are severely limited and largely unsuitable.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of hydraulic barriers.	
Pro	***Contra***
• Require little maintenance if designed correctly • Highly effective for upstream migrating fish • Often stable and benign • May be inexpensive at small scale, especially if local materials are used • May provide habitat or water quality benefits (i.e. improved DO)	• Expensive at large scale, especially if large machinery is required • Not suitable for low gradients • Ineffective for downstream migrating fish or lentic habitats • Navigational hazards • Floods may damage • Long-term maintenance costs

2.2.2 Screens, weirs and nets

Net, screen and weir barriers consist of mesh, netting, piping or other materials with interstitial pore sizes or spacing large enough to permit the flow of water, yet small enough to restrict the passage of target fish, thus physically excluding the undesirable fish.

Overwhelmingly, the most common use of screens, nets and weirs is in conjunction with hydroelectric projects to reduce entrainment in turbines (Turnpenny et al. 1998). There are many methods used to prevent entrainment of fish, and many of these have the potential to be adapted to create barriers of downstream dispersal. Reporting on the efficacy of these barriers is limited, though more extensive than other categories of barriers.

Several authors have reviewed the use and application of physical screens as they pertain to reducing entrainment of industrial activities. Turnpenny et al. (1998) review both the legislative issues surrounding fish diversion at power stations and specific exclusion technologies available in the United Kingdom. Anonymous (1995), a report from the American Office of Technology Assessment, summarize both downstream and upstream fish passage technologies with emphasis on federal energy regulations. Larinier and Travade (2002) describe the issues of downstream fish passage technology and provide examples of downstream fish passage technologies in France.

Only one review pertained to controlling downstream dispersal of non-native fishes, though again it focused on fish control applications at hydroelectric facilities. Miller and Laiho (1997) provide a feasibility evaluation of installing control structures to eliminate downstream escapement by non-native fish from reservoirs in the Colorado River basin. The three non-native species examined included smallmouth bass, northern pike and channel catfish and the authors suggest several options for both high- and low-velocity scenarios. A common conclusion of the

authors, and perhaps most pertinent to the containment of non-native fishes is that no single screen, weir or net barrier technology is suited to all conditions. The choice between static (stationary), mechanical (moving) and other screen technologies should be based on hydrological, physical and biological criterion specific to the site. Because of the large number of designs currently used, particularly at hydroelectric dams, a wide variety of options are available to fit most any biological need or environmental constraint (Turnpenny et al. 1998).

Few non-industrial uses of screen, weir or net technologies have been reported and few have been applied specifically for containment of undesirable fishes, though the potential to adapt these structures is high.

The single example found of screens deployed to limit the downstream dispersal of non-native fish was reported by Rischbieter (2000), where the author described efforts to prevent non-native northern pike escapement from Lake Davis, California. The author describes the installation of a "grater" screen to an existing power generation plant. This screen effectively dismembered all fish entrained in the turbines ensuring that all fish leaving the hydroelectric plant were fatally wounded. Eggs and larvae were not addressed in this study.

In an effort to control non-native topmouth gudgeon in an English pond, Britton and Brazier (2006) describe the use of a screen on the main outflow of the pond. However, the screen was determined to be ineffective (<100%) at preventing the spread of gudgeon as individuals smaller than 20 cm were able to traverse the screen barrier. Consequently, managers resorted to chemical reclamation of the pond (Britton and Brazier 2006).

In an effort to improve fall angling for stocked brook trout in two New York headwater lakes, Josephson et al. (2001) deployed mesh barriers at the lake outlets. The efficacy of these mesh barriers was not examined, though the authors implied they prevented emigration of larger trout.

Triploid grass carp were introduced to Lake Seminole, Florida for biological control of an invasive aquatic plant (Maceina et al. 1999). To confine these carp to two embayments, they deployed a V-shaped funnel weir and two boat-traversable gates. Radio-tracking of 119 grass carp at the V-weir site and 69 carp at the boat-traversable gate sites showed the barriers contained as few as 58% and 65% of the test fish, respectively.

Screen, weir and net barrier options to limit the upstream movement of undesirable fishes are not well explored, though presumably many of the logistical issues of screen, net and weir use for downstream barriers would apply.

Fish containment using screens, nets and weirs often fail to prevent the dispersal of juvenile fish and eggs that may survive and colonize areas outside of the containment zone, as was the circumstances described by Britton and Brazier (2006). Mesh size of nets and screens or the space between weir bars must be small enough to prevent fish from passing through, however in flowing water or in areas with large amounts of debris; nets may restrict water flow and cause undue environmental effects or destroy equipment. A company known as Gunderboom Incorporated, based in Alaska advertises a "marine life exclusion system (MLES)". This cloth-like barrier has interstitial pore sizes sufficiently small to prevent passage of eggs and larvae, yet large enough to permit water movement (Gunderboom 2009). In a study of the efficacy of a Gunderboom MLES deployed at the Lovett (Power) Generation Station, New York, the system reduced entrainment of ichthyoplankton by 80% (Raffenberg et al. 2002). While no examination of the containment of larger fish, presumably they were effectively contained. The market for

these barriers is primarily hydroelectric or other water resource facilities, however the applicability of this technology to the control of undesirable non-native fishes should be examined.

Coanda-effect screens are a promising new development. If these screens were adapted to the containment of fish, a typical set-up would be as follows. A water diversion, efficient enough to limit the movement of target-sized fish, larvae or eggs would direct water to a tilted (horizontal) wire screen. This screen is of fine pore size and shaped much like a typical playground slide. When water passes over the screen, clean water falls through the screen and fish and debris remain on top of the screen where they slide/ wash into a holding area (Turnpenny et al. 1998). The washing of fish and debris from the screen is principally based on the Coanda effect, which is the tendency of a jet stream to adhere to a curved wall (Brown 1999). Coanda-effect screens are highly effective and are inherently self cleaning (Wahl 2003), and address many of the logistical issues facing screen fish barriers in rivers subject to high debris loads (Brown 1999).

In summary, screens are effective and can completely block fish (larger than the mesh size), thus effectively addressing biological issues such as fish exclusion. However they often create engineering issues of decreased water flow and the accumulation of debris (Bainbridge 1964). Screens and nets are generally inexpensive, though maintenance and cleaning associated with screens can be intensive and costly, particularly in environments with large amounts of debris or high flow (Brown 1999, Turnpenny et al. 1998). Finally, a plethora of screen and net options exist. The application and suitability of a particular screen design depends largely on the target species to be contained, the direction of containment, the ability to provide maintenance to the system and the velocity and quantity of water at the barrier site.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of screen, net and weir barriers.	
Pro	***Contra***
• Wide range of applicability • Highly effective for specific fish sizes and life stages	• Often high maintenance • Highly size selective • Can pose navigational hazards

2.2.3 Other physical barriers

Unexplored technologies may prove to be valuable tools to prevent the movement of non-native fishes, and projects originally designed for alternate purposes may be adaptable to fish control. For example, Turnpenny et al. (1998) describe the use of under gravel intakes for hydroelectric plants. Perforated pipes laid beneath clean gravel "filter" water prior to intake by the turbines. This technology could be adapted to the exclusion of undesirable fishes. It is estimated that 40 m^2 of gravel area is required for each $1m^3 \cdot sec^{-1}$ of flow, however in stained or turbid waters, a larger area may be required or the system may fail as a whole (Turnpenny et al. 1998), presumably due to clogging of the gravel.

For downstream barriers, in particular lake outlets, habitat preferences of the target species may improve barrier efficiency. For example, smallmouth bass are primarily considered a littoral zone

fish (Pleug and Pauley 1984). In many northern lakes, smallmouth may use the deepest mid-lake habitat only seasonally, if at all. Thus, if natural outlets are dammed and an alternate, artificial intake pipe was situated in this unfavoured mid-lake habitat, downstream movements of bass may potentially be reduced. If other forms of barriers (i.e. electric barriers, screens etc.) were placed in such areas, the efficacy of the barrier may be greatly improved.

2.3 Behavioural Barriers

2.3.1 Electrical barriers

Electricity has been used in fisheries science since the late 19th century, first as a method to manipulate the movements of fish, and later as a fish sampling tool (Reynolds 1996). Water can be an excellent conductor of electricity and when a pair (at minimum) of electrodes is submersed in water, an electrical current can be transferred between electrodes. The electrical current between the electrodes travels through the water in a theoretical three-dimensional field (Reynolds 1996). The voltage of electricity decreases with distance from the electrodes and is often described in terms of equipotential lines (Reynolds 1996).

When a fish is perpendicular to these equipotential lines, it can either be immobilized or exhibit involuntary galvanotaxis (involuntary swimming). The more equipotential lines that are crossed; the greater the effect on the fish (Reynolds 1996). As such, effective electrical barriers are designed based on the minimum size of fish targeted.

Early electrical barriers used pulsed alternating currents (AC) and afflicted high mortality on all in-range fish species. Later developments successfully paired pulsed direct current (DC) barriers with the aforementioned AC currents to repel fish prior to entering the lethal zone (McLain 1957). In more recent years, managers most commonly deploy direct current systems (Swink 1999). Numerous laboratory tests have demonstrated the ability of electric barriers to deter fish (i.e. Dawson et al. 2005, Savino et al. 2001, Stewart and Copland 1976, among others) and this is mirrored in field applications, particularly when inhibiting upstream migration.

Some of the earliest reports describing field applications of electrical barriers came from the Great Lakes region as a control measure for sea lamprey (Applegate et al. 1952 *in* Clarkson 2004, McLain 1957). These projects used electrical barriers to prevent the upstream spawning migrations of lamprey into tributaries of the Great Lakes with the goal of reducing recruitment into the lamprey population (Carlson 2003). These efforts proved successful at the stream-scale but failed to eliminate the lamprey-derived mortality on lake trout; which eventually contributed to population collapse (Lawrie 1970). The electrical barrier program was abandoned when the use of lampricides gained popularity (McLain et al. 1965 *in* Clarkson 2004). From 1986 to 1995, the price of lampricides tripled and with increasing societal concerns of annual application of lampricides, the Great Lakes Fishery Commission set targets of reducing piscicide use by 2001 to 50% of the late 1980's levels (Great Lakes Fishery Commission 1995). Consequently, several type of barriers re-gained popularity, one of which was DC electrical barriers (Great Lakes Fishery Commission 2009, Swink 1999). Swink (1999) suggested that electrical barriers could be used on low-gradient streams unsuited for lowhead dams thus reducing dependence on lampricides. Swink also indicated that tests of efficacy revealed a "complete blocking of lamprey spawning runs".

Asian carp (silver carp, bighead carp, black carp) are a highly invasive non-native fishes that are now found in the Mississippi River, USA. In an effort to prevent Asian carp from entering the Great Lakes via the Chicago Sanitary and Shipping Canal and potentially devastating the US $4 billion dollar fishing industry, authorities have deployed electric barriers (Anon. 2008, Dettmers et al. 2005, Shea 2006, US Army Corps of Engineers 2009). The first phase of the barrier has been in place since 2002 and is considered effective for large carp (Shea 2006), as it effectively accommodated a wide range of fish sizes (Dettmers et al. 2005). The cost of the first phase of the project was estimated at US $4.2 million, the second phase of the project was budgeted at US $9.1 million, and annual operating costs were estimated at US $400,000 (Shea 2006).

In an effort to control the upstream spread of non-native fishes, electrical barriers were constructed in 1988 in two canals of the Central Arizona Project (CAP), a series of surficial, uncovered canals that deliver water from the Colorado River to three counties in Arizona, USA (CAP 2009; (Clarkson 2004). During barrier operations from 1988 to 2000, power outages, equipment failures and human error lead to a documented 100 hours of "down-time", representing less than 0.001% of the total installation period (Clarkson 2004). As a result, Clarkson (2004) indicated that the outages certainly allowed transgressions by undesired fish(es), an assertion supported by evidence of non-native grass carp, stocked below the barrier, above the barrier. Furthermore, grass carp were known to have transgressed the barrier during periods of uninterrupted operation (Clarkson 2004). This outlines the degree to which barrier efficiency is important for the containment of non-native fish and demonstrates that any barrier breach (by two or more fish) may potentially render the entire operation a failure.

Verril and Berry (1995) also assessed the ability of an electric barrier to limit the upstream movements of fish. Over two years, they tagged 1,600 bigmouth buffalo and common carp, released them below an electric barrier and examined fish in the lake above the barrier. Of the 3,376 common carp and bigmouth buffalo captured above the barrier, none were marked fish from below the barrier. The authors thus reported the barrier to be effective.

In an effort to control an invasive aquatic plant, triploid grass carp were introduced to Lake Seminole, Florida. To confine these carp to a restricted area, Maceina et al. (1999) deployed a V-shaped funnel weir fitted with an electric barrier at the outlet. Radio-tracking of 84 grass carp show the barriers to successfully contain 100% of the test fish, compared to containment rates as low as 58% to 65% for weir and gate structures (Maceina et al. 1999).

The use of electrical barriers to control downstream moving fishes has not been extensively explored or has been underreported. Downstream migrating fish are assisted by the flow of water and as fish approach an electrical current, they become progressively less able to escape the barrier area and are inevitably swept past the barrier in a state of taxis (Bainbridge 1964). As such, electric barriers deployed to prevent the downstream movement of fish are generally considered to be less than 100% effective under optimal conditions, while those used for the prevention of upstream movement are generally accepted to be more effective (SRI 1990 *in* Clarkson 2003).

Several important logistical considerations should be considered prior to the use of electric barriers, including water depth and the width of the barrier which can affect its efficacy (Bullen and Carlson 2003). Human activities may also limit electrical barrier deployment. Dettmers et al. (2005) showed that boat traffic can influence the effectiveness of electrical barriers as the presence of ships with conductive steel hulls increases the time required to immobilize target

fish. Furthermore, from a human safety standpoint, an electric barrier may pose significant threat. To address this, authorities at the Chicago Shipping and Sanitary Canal installed chain-link fence on the landward borders of the barrier (Anon. 2008). This of course does not address the potential of boaters falling into water and being electrocuted.

In summary, there is evidence to suggest that electric barriers can efficiently deter fish passage (i.e. Clarkson 2004) and in some cases provide a 100% effective barrier (i.e. Maciena et al. 1999). However, electric barriers alone are not likely to be 100% effective (Bullen and Carlson 2003, Turnpenny et al. 1998).

Summary of positive attributes (*Pro*) and limitations (*Contra*) of electric barriers.	
Pro	***Contra***
• Suitable for low gradient streams where hydraulic barrier inefficient • Highly effective for upstream migrating fish or those in lentic habitats if properly set-up & maintained	• High maintenance requiring access to sustained power supply • Susceptible to equipment or operational failure • Limited use for downstream migrating fishes • Limited to shallow areas

2.3.2 Bubble barriers

Bubbles emitted from a pipe or tube mounted at depth rise through the water column to create the illusion of a physical barrier and also emit an acoustic cue, both of which may repel fish (McIninch and Hocutt 1987, Patrick et al. 1985, Sager et al. 2000a, Turnpenny et al. 1998). Laboratory studies assessing the efficacy of bubble barriers often reported mixed results, with some evidence that small fish are effectively deterred by bubble barriers (Stewart 1981), and efficiencies as high as 98% have been reported (Solomon 1992 *in* Baumgartner 2005). However, other laboratory experiments have concluded that bubble curtains are of limited use. For example, in laboratory experiments using Eurasian ruffe, Dawson et al. (2006) showed that air bubble curtains significantly repelled ruffe but were of limited use for stopping movements of ruffe, as ruffe continued to successfully traverse the barriers at an average rate of 4.5 times/fish.

In other laboratory experiments using common mid-Atlantic estuarine species such as white perch, spot or menhaden, bubble curtains alone did not effectively function as a barrier (McIninch and Hocutt 1987, Sager et al. 2000a). McIninch and Hocutt (1987) reported that rates of deterrence did not exceed 50% and in some cases, bubble curtains attracted white perch (McIninch and Hocutt 1987).

Field experiments also generally reported low efficiencies and on average, optimal efficiencies of bubble curtains have been estimated at about 40% (Turnpenny et al. 1998), though some studies have indicated a complete lack of efficacy (Lieberman and Muessig 1978). A potential explanation for the discrepancy in effectiveness between laboratory tests and field applications is that under the cover of darkness, fish were less likely to be deterred. When air bubbles were illuminated with strobe lights, rainbow smelt, gizzard shad and alewife were repelled at 90% to

98% efficiency (Patrick et al. 1985). Similarly, McIninch and Hocutt (1987) showed that bubble curtains illuminated by strobe light increased rates of deterrence to almost 75% for the three aforementioned estuarine species.

In the context of containing 100% of undesirable fishes, there is sufficient evidence to indicate that bubble curtains alone are not effective. However, reports of significant intra-specific variation of the efficacy of bubble barriers suggests that additional research may support field applications.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of bubble barriers.	
Pro	***Contra***
• Inexpensive • Low maintenance • Benign • Well suited to slow flowing water	• Reported < 100% effective or entirely ineffective • Largely untested • Highly species specific • Not suited to fast flowing water

2.3.3 Acoustic barriers

Acoustic barriers emit sounds within the audible range of the target species to cause avoidance behaviours, either due to auditory discomfort or the perception of the noise as a threat. Unlike electric stimuli, fish have the ability to easily locate the source of acoustic stimuli, and thus have the ability to avoid that source (Bullen and Carlson 2003).

Early studies on the effectiveness of acoustic deterrents were primarily focused on Clupeidae such as blueback herring and alewife which were shown to exhibit high avoidance behaviour when subjected to sounds of the proper frequency (Haymes et al. 1986, Nestler et al 1992). These species would later highlight the potential for acoustic fish impediment. In a study of the effectiveness of an acoustic deterrent system at the James A. FitzPatrick Nuclear Power Plant, on the shore of Lake Ontario, New York, Ross et al. (1993) showed that alewife impingement rates in the intake screens were reduced by up to 87% using acoustic barriers.

Several field experiments on other fish species have produced contradictory results which are generally attributed to species-specific differences in abilities to detect acoustic cues, as summarized in Bullen and Carlson (2003). For example, during an examination of the ability of acoustic deterrents to reduce impingement of fish into a Belgian power plant, Maes et al. (2004) reported a mean impingement reduction of 60% (range 38% to 95%) for fish species of the estuary. Reductions in impingement were highly species-specific, which they explained as most probably a result of differences in detection and swimming ability, though most species with swim bladders exhibit some sort of avoidance behaviour (Maes et al. 2004).

Confounding effects also occurred for closely-related species of similar physiology and even within a single species. Carlson (1994) provides a summary of acoustic barrier principles and a history of their use associated with power plants. He indicates that acoustic deterrents can effectively reduce impingement of many fish species to hydroelectric dams, although salmonids appear to be more difficult to deter and successful programs for pacific salmon had not yet been

demonstrated. Knudsen et al. (1994) showed that downstream migrating Atlantic salmon smolts were effectively deterred from previously-used stream channels using infrasound emissions. Knudsen et al. (1997) indicate that Chinook Salmon and Rainbow Trout showed avoidance behaviour to an acoustic signal. Unlike other authors, but similar to their earlier studies, they used a low frequency signal (infrasound), to which they attribute the deterrence response and suggest that acoustic deterrent devices may be a valuable technique to reduce entrainment.

Conversely, in cage tests of the response of chinook salmon and northern pikeminnow to infrasound deterrents, Amaral et al. (2001) concluded that chinook salmon did not respond to infrasound, however northern pikeminnow, a significant predator of Chinook salmon smolts, strongly avoided the acoustic signals. The conclusion of Amaral et al. (2001) were that acoustic deterrents may be used to repel northern pikeminnow from areas of high-density Chinook salmon smolts, and thus reduce total loses to predation. In contrast, Mueller et al. (2001) indicated that while chinook salmon showed that some behavioural response when subjected to infrasound, the closely related species, brook trout and rainbow trout, were not affected. A possible explanation for these contradictory results may be the duration to which chinook salmon were exposed to acoustic cues, as is it known that over time, fish may become acclimatized to acoustic deterrents (Bullen and Carlson 2003, Carlson 1994, Mueller et al. 2001). As such, short term studies would conclude that fish avoided the signals while longer-term studies would show a gradual loss of signal-avoiding behaviour.

Bullen and Carlson (2003) indicate that "*although infrasonic acceleration fields do induce fright reactions in fish, these reactions are often not of sufficient strength to dissuade fish from passing through a sound field when a positive stimulus is introduced*". In other words, if fish have sufficient motivation to enter an area, they likely will, regardless of the presence of an acoustic deterrent. As such their application to projects requiring 100% exclusion appears limited. In fact, acoustic deterrents may not be effective barriers on their own (Burner and Moore 1953 *in* Bainbridge 1964). An example of the success of multiple barriers was illustrated when Atlantic cod were contained more efficiently by paired electric plus acoustic barriers than either barrier type individually (Clegg 1997 *in* Bullen and Carlson 2003).

A common conclusion among authors is that acoustic deterrents were still largely under-explored and that further research is required prior to wide-scale deterrent applications or deploying acoustic barriers for the absolute containment of fishes. Such investigation may include the physiological features that preclude species sensitivity to acoustic signals as well as environmental determinants of successful acoustic deterrent operations. For example, turbidity is known to have little effect on acoustic integrity (Anon. 1995); however, variables known to influence sound transmission in water, such as thermoclines (Carlson 1994), turbulence and the presence of solutes, are likely factors that will modify the effectiveness of acoustic barriers.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of acoustic barriers.	
Pro	***Contra***
• Inexpensive • Low maintenance • Highly species specific	• Reported < 100% effective • Largely untested • Highly species specific

2.3.4 Light-based barriers

Many fish are highly responsive to visual stimuli, and light may alter fish behaviour (Whitney 1969). High intensity, flashing light or light perceived as "noxious stimuli", may elicit avoidance behaviour in fish (Bullen and Carlson 2003, Popper and Carlson 1998). The most common light-based barriers involve strobe (flashing) lights, though continuous lighting is also used (Turnpenny et al. 1998). Several projects have deployed strobe lights to deter or attract fish (Bullen and Carlson 2003).

As a proxy to stress associated with subjection to flashing strobe lights, Sager et al. (2000b) measured respiration rates of white perch and spot. They reported increased respiration rates when fish were subjected to flashing strobe lights, particularly in dark conditions and thus concluded that strobe lights stress white perch and spot and may potentially be used as a deterrent.

Laboratory studies examining the efficacy of light-based deterrent systems reported mixed and sometimes contradictory results. In a study on white perch, spot and menhaden, Sager et al. (2000a) concluded that strobe lights may be potentially used as a deterrence system, especially when strobe lights were paired with bubble curtains, but that species and environment need to be considered. Furthermore, they discovered that at some strobe settings, all three species exhibited some degree of attraction to strobe lights. These results have been replicated with other species. Nemeth and Anderson (1992) examined the response of juvenile coho and chinook salmon to strobe and mercury vapour (non-flashing) light under laboratory conditions. They concluded that both species avoided strobe lights and mercury vapour lights at full intensity, however chinook salmon were attracted to a low intensity mercury vapour light. Also in laboratory settings, American eels were shown to be repelled from a strobe light at efficiencies as high as 94% (Patrick et al. 1982).

Field application and evaluation of light-based barriers is less extensive. Other than field-based experiments on American eels, no study reports in-situ efficiencies of light-based technologies as barriers. In a follow-up to their laboratory studies with American eel, Patrick et al. (1982) examined the efficiency of a white strobe light to deter eels from a previously used fish ladder. They concluded that the strobe light deterred 65% to 92% of upstream migrating eels.

Caged smallmouth bass and northern pikeminnow were exposed to strobe lights and chinook salmon were exposed to both strobe light and "drop lights" (Amaral et al. 2001). Smallmouth bass and chinook salmon exhibited strong avoidance of strobe light during night-time trials, however northern pikeminnow did not exhibit similar avoidance. None of the three species were deterred during daytime and lowlight conditions. Amaral et al. (2001) also showed that chinook salmon exhibited only minimal avoidance behaviour to drop light.Maiolie et al. (1999) evaluated the response of kokanee salmon to strobe lights lowered into the water column under natural conditions at Dworshak reservoir, Idaho. They found that kokanee salmon were successfully repelled 20m to 40m from the light source. Additionally, after nearly 6 hours, no kokanee ventured closer than 24m to the strobe light. Though not specifically testing a barrier, their experiment revealed the response of wild fish to in-situ strobe light stimuli.

Light-based barrier technologies hinge on the premise that fish view the light and elicit an avoidance response. As such, some researchers suggest that the effectiveness of light barriers depends largely on the ability to transmit light through water (Sager et al. 2000a). It would therefore seem logical that water turbidity would negatively correlate with the effectiveness of

light-based barriers, however a positive relationship has been demonstrated (McIninch and Hocutt 1987).

The response of fish to light-based barriers varies significantly among fish species, from high to low deterrence (Amaral et al. 2001, Sager et al. 2000a) and even attraction (Nemeth and Anderson 1992).). Furthermore, considering that response is dependent on source, intensity and pattern of light (Amaral et al. 2001, Nemeth and Anderson 1992), additional studies prior to wide-spread application of light-based technologies as barriers to migration are required.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of light-based barriers.	
Pro	***Contra***
• Inexpensive • Require little maintenance • Highly species specific	• Largely untested • <100% effective • Potential for behavioural accommodation • Highly species specific

2.4 Summary of Barrier Options

Several reviews provide a good synopsis of the barrier technologies, primarily in the context of fish passage and preventing entrainment of fish at hydroelectric stations and water intake facilities (Anonymous 1995, Larinier and Travade 2002, Turnpenny et al. 1998). However, only the review to place fish barrier technologies in the context of restricting the movement of non-native fishes was that of Miller and Laiho (1997). A summary of the advantages, disadvantages and properties of each method is shown in Table 1. Commonly, these reviewers suggested that no single barrier solution will apply to all sites or all species and that barrier selection be based on site-specific information. As a consequence, they often indicate that cooperation among biologists and barrier designers is crucial. Another common conclusion is that the deployment of more than one barrier may increase overall efficiency and increase the potential for 100% containment/diversion. By constructing multiple barriers in several areas, fish that breach one barrier are likely to be contained, if only temporarily, by the successive barrier. Thus, it has been suggested that these incursions could be isolated and potentially managed prior to further dispersal or establishment (Carpenter and Terrell 2005, Meffe 1983).

Several authors have also suggested that efforts to completely block the movements of undesirable fishes may require a multi-faceted approach using several technologies at one location. For example, bighead carp, an invasive fish of the Great Lakes, were deterred by combining bubble, sound and electric barriers (Taylor et al. 2003 *in* Dawson et al. 2006). Bullen and Carlson (2003) examined the use of acoustic and electric barrier in unison and concluded that the system had potential to control large salmon with high efficiency (90%-95%) and that the paired system was superior to the relatively ineffective acoustic-only barrier.

While behavioural barriers are promising, they remain largely untested. Additionally, behavioural barriers are not likely to perform as well as physical barriers across all hydraulic

conditions (Anon. 1995) and heightened motivation for movement (i.e. spawning cues, crowding) or extreme environmental conditions (i.e. floods, drought) may reduce the efficacy of behavioural barriers (Dawson et al. 2006).

Finally, a unanimous conclusion of most authors is that behavioural barrier are less effective than physical barriers and physical barriers consistently offer the highest efficiency. For example, in the United Kingdom, when commercial proponents apply for hydroelectric or water withdrawal permits, the majority of fisheries authorities mandate the use of physical screens rather than behavioural barriers due to their reputation for more certain fish diversion (Turnpenny et al. 1998). It has been stated however that the efficiency of physical barriers may be limited by logistical constraints, such as the removal of debris, adaptations to the site and biological requirements (Anon. 1995). Furthermore, it has been documented that physical barriers (screens) can cause minor injuries (Neitzel et al. 1988), though for the purpose of restricting the distribution of non-native fishes, physical harm to fish is not a large concern unless a large number of native fishes also contact the barrier.

3.0 CONTROL AND ERADICATION OF NON-NATIVE FISHES

3.1 Introduction to Control and Eradication of Non-Native Fishes

The containment of undesirable fishes may prevent range extension and further dispersal, however containment is generally considered to be a temporary management action. Barriers are expected to only retard the spread of undesirable fishes (Carpenter and Terrell 2005). Furthermore, containment does not address issues associated with the presence of undesirable fishes in areas already colonized. As such, authorities charged with managing undesirable and non-native fishes often attempt to manipulate the population parameters of the target fish. Such parameters may include abundance, population size structure or growth. The manipulation of target fish population may reduce predation on native biota (Bestgen et al. 2007, Ogutu-Ohwayu 1993) or reduce competition with native fishes (Baker et al. 2008, Bohn and Amundsen 2001).

The process of managing undesirable fishes, whether deliberate or not, most often answers three basic management questions;

- Is control or eradication warranted?
- Is control or eradication possible? And
- What management method provides the best opportunity to achieve management goals?

Beamesderfer (2000) suggested that public acceptance was also an important consideration. In general, management programs that employ techniques presumed to threaten human health generate the most public opposition, as is the case with chemical piscicides. For the scope of this section, public acceptance will not be discussed in detail.

3.1.1 Is control and / or eradication warranted?

This first management question addresses the level of risk associated with the presence of the non-native fish and is most often disseminated in the form of a risk assessment. The ability of the species to survive, reproduce and colonize the area is considered. The risk of geographic

population expansion (spread) may also be addressed. Finally, a risk assessment allows determination of potential threats to native biota, particularly in the form of competition, predation and genetic introgression. If the potential of a non-native fish species to threaten native biota is considered high, then managers may wish to explore control and/or eradication options. Thus, the proceeding question may be warranted.

3.1.2 Is control and / or eradication possible?

The ability to control and/or eradication undesirable, non-native fishes is influenced by the potential to affect change via management actions (Beamesderfer 2000). The potential for management will be bound by several factors pertaining to the biology and ecology of the target fish, the environmental/ habitat-related constraints of the habitat being colonized and the potential for re-invasion or re-introduction.

The management decision making process was examined by Chadderton (2003) with particular reference to non-native brown trout in New Zealand. These non-native fish posed a significant threat to native biota (Townsend 1996) and local fisheries managers were faced with the decision to manage the non-native fish or accept the species. The author provides the following eradication criteria to assist in determining the management feasibility.

> *"All must be met before eradication should proceed*
>
> *1. Rate of removal exceeds rate of increase (recruitment). Links to other criteria. Eradication will only succeed if this is achieved.*
>
> *2. Low probability of reinvasion. Requires assessment of vectors, source of founder populations, dispersal pathways. Linked to 5. - potential for people to*
>
> *reintroduce.*
>
> *3. Ability to target all individuals in a population, either directly or by preventing recruitment (i.e. removing all individuals before they reach sexual maturity).*
>
> *4. Populations can be monitored and targeted at low densities. Target species needs to be able to be detected at low densities otherwise survivors may re-establish the infestation*
>
> *5. Suitable socio-political environment. Eradication objectives and methodology require local and national support.*
>
> *6. Commitment /certainty. Appropriate staff and resources need to be accurately identified and committed for the length of the program."*

The principles upon which Chadderton (2003) compiled these criteria are applicable to other undesirable fish species and may be considered when planning their management. The ability to remove fish at a rate greater than recruitment, the ability to target all individuals and the ability to target fish at low densities are all contingent on the control or eradication method.

3.1.3 What action provides the best opportunity to achieve the management goals?

As previously discussed, the removal of individuals in order to reduce abundance may be termed control, while removal of 100% of all life stages of a target population is termed eradication.

Control and eradication efforts often rely on similar capture techniques but most frequently differ in intensity and / or duration. Methods for control and eradication can most generally be classified into four categories:

- Mechanical: involves physically removing individuals using nets, traps, electricity, explosives or angling. Fish are captured or injured.
- Chemical: chemicals deployed to kill fish (piscicides).
- Biological: biological removal of undesirable fishes is, in the most general sense, the manipulation of fish, their ecology or interactions of fish and other organisms to result in decreased abundance of the target fish.
- Environmental: involves the manipulation of fish habitat to create conditions that limit growth, reproduction or survival.

The choice of an appropriate control or eradication technique depends largely on the biology / ecology of the targeted fish species, the type of habitat being managed, logistical constraints such as funding or access, public perception and the management program objectives. Arguably, the most important and first step to selecting an appropriate control or eradication technique is the definition of the management objectives. By matching the efficacy of removal techniques (or potential efficacy of untested removal techniques) to the desired outcome, managers may select a removal technique that provides the best opportunity for success. Measures of success must also be identified prior to treatment. Many parameters have been identified for the evaluation of fish removal projects including reductions in proportional stock density (PSD), reduction in abundance, reduced competition / predation, and shifts in size structures of either target or native fishes (Meronek et al. 1996).

Managers should also have a sound understanding of the biology and ecology of the target fish. Life history may limit removal options and emergent population-level traits may buffer removal efforts. For example, Zipken et al. (2008) reported that following intensive removal efforts juvenile smallmouth bass recruitment increased due to increased survival resulting from decreased competition.

To further guide the selection of appropriate removal techniques, collateral effects on non-target biota must be considered, especially in environments containing ecologically important, sensitive, rare or valued species. Efforts to limit the impact of removal efforts can in some cases increase the attractiveness of otherwise detrimental methods. For example, managers in Arizona implemented an elaborate and extensive native fish salvage and repatriation program associated with a chemical stream reclamation project that would otherwise have eradicated the native fish, some of which were considered rare (Weedman and Sponholtz 2005).

3.2 Mechanical Technologies

3.2.1 Electrofishing as a control technology

Similar to electric barriers, electrofishing as a removal technique is based on the reaction of fish to an electrical current. When a fish crosses an electrical current in water, the fish may be "stunned" (known as tetany), a result of involuntary muscle spasms (Reynolds 1996). The use of electrofishing gear for the sampling of fish has been reported by fisheries scientists as early as

1931 (Burr 1931). Presently, the science of electrofishing is well advanced and the physiological effects of electrofishing on fish extensively evaluated (Reynolds 1996).

The application of electrofishing as a sampling technique is similar across most habitats, though the gear may vary. Reynolds (1996) provides a good summary of electrofishing systems, from which the following brief table describing common gears is adapted.

Habitat	Gear Type	Advantages
Wadeable streams	Backpack electrofisher	• Easily transported • Permits access to confined habitats
Navigable streams, rivers and lakes	Boat- or barge-mounted electrofisher	• Samples large area • Samples deeper water
Littoral zone	Electric seine*	• Highly effective for small fish • Highly effective in shallow water

* see Bayley et al. (1989), Hawkins et al. (2008)

Using one or more gear-types, electrofishing has been successfully used to remove non-native fish in both lentic and lotic habitats. For example, in several US national parks and national forests, non-native trout seriously threaten native stream salmonids through inter-specific competition, disease and predation. In an effort to increase native salmonid population viability, programs using backpack electrofishing gear to remove non-native trout species were initiated. Baker et al. (2008) describe a program in Great Basin National Park where rainbow trout were removed from a very small brook (discharge ~0.05 $m^3 \cdot sec^{-1}$) using backpack electrofishing gear. A crew of three fished the brook at least twice a year from 2002 to 2005. Rainbow trout were at low density (estimated at 1.2 per 100 m^2), and after removing a total of 20 trout in the first three years, the brook was considered void of non-native species. In contrast, Thompson and Rahel (1996) provided an evaluation of a two-year non-native brook trout removal project using backpack electrofishing gear in three very small Wyoming streams (discharge = 0.02 – 0.04 $m^3 \cdot sec^{-1}$). They estimated that 73 - 100% of young-of-year (0+) brook trout were removed and 59 - 100% of older trout (1+ and older) were removed and that eradication did not occur.

Moore et al. (1983) removed non-native rainbow trout to assess the response of sympatric native brook trout. Four 2nd or 3rd order streams in Great Smoky Mountains National Park were targeted with backpack electrofishing gear. After four years of variable removal efforts, rainbow trout standing stocks had been reduced by 83% to 94%, again, failing to eradicate the target fish. Later efforts, summarized by Moore et al. (1986), of a similar non-native rainbow trout removal in five small streams (discharge <0.5 $m^3 \cdot sec^{-1}$) using backpack electrofishing gear resulted in 42% of the young-of-the-year and 55% of the older trout being removed after the first year of the project (three passes). At the end of the six year project, the average population reduction for all ages of rainbow trout across the five streams was 89% (Moore et al. 1985). Kulp and Moore (2000) later re-evaluated the potential for 100% eradication, and found that five, three-pass depletion removals were required for 100% eradication in small streams.

Rinne (2001) described a non-native predatory fish removal program on the Verde River, Arizona that used backpack electrofishing gear. While the removal of green sunfish and yellow bullhead showed mixed success, the removal effort reduced the smallmouth bass populations by 50% to 80%.

Shepard et al. (2002) described an eight year brook trout removal effort in a small stream of simple habitat where brook trout were, at last survey, successfully eradicated. The authors indicated that removal of in-stream cover, such as woody-debris and overhanging vegetation, significantly enhanced their electrofishing efficiency. Furthermore, the use of a fish barrier in the lower part of the stream (for upstream migrating fish) was considered an integral part of the project as it prevented recolonization by brook trout from other areas.

Earle and Lajeunesse (2007) reported a failed attempt to remove non-native brook trout in a small Alberta creek (discharge = $0.26\ m^3 \cdot sec^{-1}$). Over 2,500 brook trout were removed with an electrofishing effort of five, two-pass depletion removals (and another 5,600 0+ trout by trapping), resulting in immediate decreased brook trout biomass and CPUE. Only four years after the end of intensive removal, densities had returned to or exceeded pre-removal levels.

Efforts to remove fish via electrofishing in larger streams are limited, though there have been a few reported projects. Burdick (2008) reported efforts to remove smallmouth bass and four other centrarchid fishes from over 160 km of the upper Colorado River (discharge ~ $125 m^3 \cdot sec^{-1}$). Using one or two boat electrofishers, river sections were fished a minimum of three times a year for three consecutive years. A large number of smallmouth bass were removed in years one, two and three (1,186, 1,596 and 836, respectively), though CPUE was variable across all three removal years, which differs substantially from successful trout removal projects conducted in small streams (see above). Based on a population estimate in the final study year, the authors estimated the smallmouth bass population prior to removals and estimated that the population had declined about 60% over the course of the removal project. However, immigration and emigration likely occurred as treatment sections were not isolated.

In an effort to control non-native smallmouth bass in another part of the Upper Colorado River basin, Hawkins et al. (2008) described the use of boat electrofishing equipment and electric seines to treat a 47 km section of the Yampa River. Sampling with a boat occurred four to ten times a year during 2003 to 2007 and sampling with an electric seine occurred during July and August, during 2005 to 2007. A total of 15,190 smallmouth bass were removed using the boat electrofisher and an additional 18,166 smallmouth were removed using the electric seine. The authors estimated that annual rates of removal were between 40% and 65%, although the resulting population change as a result of removal efforts was estimated at only 17% to 19%. The authors indicated that recruitment from adjacent areas may have supplemented the population, and density-dependant survival may have counteracted the efforts of large-scale removals.

Reports detailing the use of electrofishing gear in lakes, specifically as a removal technique, are scarce. In one example, Boucher (2008) described efforts of the Maine Department of Inland Fisheries and Wildlife to control smallmouth bass in a small (70 ha) pond. Following construction of a hydraulic barrier between the pond and the expected invasion source (a downstream lake), the efficacy of removing smallmouth bass using raft electrofishing gear was assessed over two years. Although CPUE was high (20.0 to 23.7 bass/hour), early evidence suggests that smallmouth bass were not eradicated from the pond. A total of 8.6 hours

electrofishing caught 200 bass the first year and another 7.3 hours of electrofishing caught 147 bass in the second year, most were two years old or younger.

A more intensive removal effort of non-native smallmouth bass from a large (271 ha) oligotrophic lake in New York was reported by Lepak et al. (2006) and Weidel et al. (2007). From 2000 to 2002, over 28,000 individuals were removed using electrofishing gear, resulting in a 90% reduction in smallmouth bass abundance. An interesting conclusion by Weidel et al. (2007) is that the smallmouth bass population was resilient to removal and that strong year-classes were produced throughout the removal program.

There is also indirect evidence questions the potential use of electrofishing gear in lakes for non-native control programs. For instance, Bayley and Austin (2002) estimated catchability (q) for ten fish species using boat electrofishing gear in near-shore (littoral) areas and open water (deep water) areas. In their study, largemouth bass, a highly littoral predator, exhibited the highest q at only 0.14. This suggests that electrofishing may be used to sample large numbers of some fish species, although the effort required for 100% removal may be immense in lakes given the modest q. An interesting conclusion of Bayley and Austin (2002) was that catchability increased with length of the fish, an observation consistent with general electrofishing theory (see Reynolds 1996). This suggests that electrofishing may bias catch, yet in the context of non-native control, allow large and potentially mature individuals to be targeted.

Since removal efficiencies for small or juvenile fish are generally lower than for large fish (Kulp and Moore 2000, Moore et al. 1986, Earle and Lajeunesse 2007), programs that effectively remove large individuals would fail to remove small individuals. Removal efforts would then be required for several years with the remaining young fish being removed as they grow to vulnerable sizes, preferably prior to maturity. Multiyear programs would likely be most effective for species with size-at-maturity that is larger than the minimum size effectively captured by electrofishing gear. Furthermore, diel trends in q would need be addressed to maximize electrofishing efficiency.

In summary, for the purpose of reducing abundances of undesirable fishes, electrofishing technologies may be suitable, though the size of area being treated, water depth, habitat complexity and other site-specific factors will modify the effectiveness. In all but the smallest streams, the suitability of electrofishing to eradicate a fish species has yet to be demonstrated. As a result, electrofishing technologies are most probably not suitable if project objectives include 100% eradication of a target fish.

Commonly, authors concluded that increased electrofishing efforts would increase the likelihood of control or eradication (Burdick 2008, Earle and Lajeunesse 2007, Hawkins et al. 2008), and that extending the duration of removal projects would likely achieve the desired control/eradication objectives (Hawkins et al. 2008). Accordingly, non-native control / eradication programs using electrofishing would need to be long-term projects requiring intensive sampling efforts and substantial financial commitment.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of electrofishing removal techniques.	
Pro	***Contra***
• Selective for fish • Proven technology • Proven effective in small stream • Few regulatory hurdles • Public acceptance high	• Required multiple efforts spanning multiple years (labour intensive) • Not well suited for complex / deep habitats • Not likely to succeed in rivers, lakes or large streams • May be selective for large individuals only

3.2.2 Netting and trapping as a control technology

Net and trap technologies have been extensively used in food fisheries and as fisheries sampling gear. However, the first reported deployment of nets and traps for the purpose of managing non-native fishes was in 1891 (Smith 1894 *in* Cole 1904) when seine nets were used to eradicate non-native common carp from a lake in San Francisco California.

Nets and traps can coarsely be divided into entanglement gears and entrapment gears (Hubert 1996), with the latter as both passive and active types. Entanglement nets use mesh fabric or material to ensnare or entangle fish that come in contact with the device (Hubert 1996). A gill net is a classic example of entanglement gear.

Entrapment gears contain fish in a closed area. The manner in which fish enter the contained area provides the major division between passive and active entrapment. In passive entrapment, fish are funnelled into an enclosed area and as a result of V-shaped or funnel structures, fish are unable to find an exit pathway (Hubert 1996). Fyke nets and Oneida (box) traps are two of the more commonly used passive entrapment gears. Conversely, active entrapment gears are moved to encircle or "sweep" fish, and are mobile in space and time (Hayes et al. 1996). Towed (trawl) nets and seine nets are two of the more commonly used active entrapment gears.

Not all species are caught with equal efficiency in a given net or trap and projects describing the use of nets / traps to remove non-native fish often report varied results, largely due to species-specific capture efficiency and environmental constraints. For example, many species of fish remain in localized habitats, often closely associated with structure (i.e. many centrarchid species in shallow littoral zone structure) and are, as such, difficult to catch using either passive or active netting/ trapping (Hayes et al. 1996).

Most netting and trapping efforts to manage non-native fishes involve passive netting. Reported efficiencies are variable and management results are often mixed. For example, in efforts to shift the size structure of northern pike populations in several small Minnesota lakes (<200ha), trap nets were found to be ineffective at capturing pike in sufficient numbers to shift the population size structure (Goeman et al. 1993). Conversely, northern pike in a 25 ha Nebraska lake were effectively captured using fyke nets, reducing biomass by 75% over 4 years and shifting the

population to smaller, younger individuals (Jolley et al. 2008). Similarly, fyke nets were demonstrated to be effective at capturing northern pike in the Yampa River, Colorado and the authors suggested that fyke nets were effective in removing pike from localized areas (Finney and Haines 2008).

Using monofilament gill nets, Knapp and Matthews (1998) were able to eradicate non-native brook trout and rainbow trout from a 1.6 ha high mountain lake in just two years of effort. The authors suggested that gill nets were a viable eradication tool for approximately 15-20% of lakes in the Sierra Nevada mountain range where non-native trout had been introduced. In a similar sized (2.1 ha) Albertan lake, brook trout were successfully eradicated over three years using gill nets (Parker et al. 2001).

Conversely, a large-scale three year gill netting effort to eradicate non-native lake trout from a lake in Yellowstone National Park, Wyoming, successfully captured 15,000 lake trout, resulting in decreased CPUE, though the authors indicate that netting had not achieved the desired population-wide effect (Bigelow et al. 2003).

Neilson et al. (2004) reported an intensive non-native Rudd removal program where gill nets of three mesh sizes were deployed in New Zealand Lakes. CPUE was reduced between 89% and 93%, however, post-removal surveys indicated that rudd populations continued to persist.

Also unsuccessful were the efforts of Earle and Lajeunesse (2007) who used unconventional traps made of 21 plastic drink containers in addition to electrofishing in a non-native Brook Trout removal program. Though they were able to remove 5,600 juvenile (<13 cm) brook trout over a period of four years, they were unable to reduce brook trout abundance.

The use of active netting and trapping is much less common in the management of non-native fishes. In a large-scale effort to reduce the abundance of non-native cyprinid species in the Green and Colorado Rivers, Utah, seine nets were used to mechanically remove undesirable species from shallow "backwaters" (Trammel et al. 2004). The authors suggested that their efforts were sufficient to reduce non-native cyprinids on a site level, though the effects of seining on a local or system-wide level were likely negligible and local sites were rapidly repopulated from adjoining, non-treated areas.

Capture efficiencies for traps and nets may also vary seasonally (Wright 2001). For example, Oneida-style trap nets have been used in Maine for sampling non-native smallmouth bass (i.e. Boucher 2005, 2006). Their efficacy fluctuated seasonally, with the period from spring ice-out until the commencement of spawning activity proving to be most effective, and post-spawn through the warmwater season providing only meagre catches (D. Boucher, Biologist, Maine Dept. of Inland Fisheries and Wildlife, pers. comm.). Wright (2001) concluded that catches of smallmouth bass in traps peaked mid-summer and that catch rate may be related to water temperature. Furthermore, Wright (2001) indicated that trapping was selective for large fish in the month of May. Based on the potential for fluctuating efficiencies, the use of several control strategies in addition to netting may be warranted, especially for season-long control programs.

The Great Lakes sea lamprey control program is an example of a successful large scale trapping initiative. Trap nets are deployed seasonally when capture efficiencies are highest. In 1995 alone, 73 tributaries were fished and the agencies captured a total of 73,415 sexually mature sea lamprey (Great Lakes Fishery Commission 1995), from which the females were destroyed and

males were sterilized and returned to the water (Great Lakes Fishery Commission 2009). Annual removal rates have been reported to be 23% to 79% (Great Lakes Fishery Commission 2001).

Traps offer the potential for selective removal of fish, and desirable fishes can separated from target fish, then released. Alternatively, traps can be modified to select only certain fishes. An example is the "Williams cage", constructed by fisheries managers in Australia as a way of removing non-native common carp (Stuart et al. 2006). The Williams cage exploits a trait common in carp where captured carp will jump in an effort to escape confinement. An evaluation of the trap indicated that it successfully separated 88% of the carp, while 99.9% of the native, desirable fish were able to avoid the trap.

Nets and traps can be an effective means of managing undesirable non-native fishes. However, the type of net / trap and effort required to achieve management objectives are determined by the biology of the target species, environmental factors affecting the capture efficiency and the sensitivity of non-target fishes to control efforts.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of netting and trapping removal techniques	
Pro	***Contra***
• Widely available gear • Relatively benign • Public acceptance high • Can be species- or size-selective	• Low efficiency • Requires intensive efforts spanning multiple years (labour intensive) • Can impact non-target fish

3.2.3 Explosives

The effects of explosives on fish and other aquatic organisms have been studied for much of the 20th century. Concussion sampling using dynamite to sample fish was first reported in 1913, (Forbes and Richardson 1913 *in* Bayley and Austen 1988). When detonated underwater, explosives emit a shock wave of positive pressure followed by rarefaction (negative pressure), both of which can cause physical harm to aquatic organisms (Cotterell 1999). The vulnerability of fish to explosives is largely species specific (Lewis 1996). Fish with swim bladders are significantly more susceptible to explosives than those without, often an order of magnitude more sensitive (Baxter II et al. 1982). In general, fish are more susceptible to the effects of explosives than other aquatic organisms (Lewis 1996).

Several type of explosives have been used including TNT, primacord, blackpowder, and several other explosive compounds (as reviewed by Lewis 1996), though only primacord (detonation cord) has been recently reported for fisheries projects (Bayley and Austen 1988, Metsger and Shafland 1986). Detonation cord is more commonly used than dynamite, due primarily to its convenience (Bayley and Austen 1988).

Reports on the use of explosives in fisheries management are not abundant, and reports on the efficacy of explosives are even less common. One example is provided by Munther (1970) who describes using detonation cord to sample smallmouth bass in the Middle Snake River, Idaho, as

a secondary sampling method to electrofishing in deep water. Earlier examples have been cited, though difficult to obtain (see Keevin et al. 2002).

In one of the most prominent papers discussing the use of detonation cord in lentic systems, Metsger and Shaffland (1986) evaluated the response of several fish species to underwater detonations. All fish within seven metres of an explosion were killed, and 88% of fish at nine metres were killed. The authors concluded that detonation cord was a viable alternative to piscicides for sampling fish.

Keevin et al. (2002) describe the use explosives to sample fish from a deep, slow section of the Mississippi River. Explosives provided a substantially larger sample of fish, containing 71% more species than did trotlines, trammel nets and hoop nets (combined) set for 24 hours. The authors concluded that explosives were a viable sampling technique when paired with hydroacoustic surveys.

Using drainable, experimental enclosures Bayley and Austen (1988) compared the efficacy of detonation cord and rotenone as sampling methods with particular emphasis on the retrieval of specimens. They concluded that fish retrieval was greater for rotenone that detonation cord, and they attributed the difference in retrieval to the inability of detonation cord to kill all fish in the enclosures.

While large fish may be affected by underwater explosions (Bayley and Austen 2002, Keevin et al. 2002, Metsger and Shaffland), the effects of explosives on small fish and eggs remains largely unknown. Research by Settle et al. (2002) and Giovani et al. (2003) examined the trauma experienced by larval fish when subjected to experimental blasts. They found that many larval fish were either dead or moribund and that commercial blasting had the potential to kill large numbers of larval fish. In laboratory experiments, eggs were shown to be rather resistant to blasts, with mortalities less than 30% in all but the most rigorous simulations (Faulkner et al. 2008).

Fish population extirpation as a result of explosive detonation is reported (Hubbs and Rechnitzer 1952 *in* Lewis 1996). The authors indicate that the area was repopulated several months later, though Lewis does not indicate to what extent Hubbs and Rechnitzer assessed fish populations.

Explosives are capable of sampling or removing fish, however they are often considered destructive to habitat, dangerous, and generally inferior to other sampling or removal methods (Hayes et al. 1996). Furthermore, the use of some explosives may leave a legacy of toxic residues, though they exhibit low bioaccumulation potential (Lotufo and Lydy 2005). Additionally, the presence of target fish in the eggs stage may complicate removal efforts via explosives.

In conclusion, explosives can kill fish although other methods are superior. Presumably, with proper planning and an excess of explosives, a large portion of fish in a system could be removed. However, several authors indicated that detonation cord should not be used when other sampling methods are applicable (Bayley and Austen 1988, Hayes et al. 1996).

Summary of positive attributes (*Pro*) and limitations (*Contra*) of explosives	
Pro	***Contra***
• May be economical in small areas of simple habitat • Can be effective if effort is intense	• Impacts non-target organisms • Destructive • Human hazards • Potential residual chemicals • Generally effective for control only

3.2.4 Angling

Angling is commonly used for fisheries sampling when non-quantitative samples are required (Hayes et al. 1996). Angling has been used to collect scientific specimens of commonly introduced or invasive fishes (Cooke et al. 2003, Hanson et al. 2007, Pflug and Pauley 1984).

Angling as a removal technique can be divided into two categories; liberalized regulations and directed angling. Liberalized angling regulations rely on an existing angling community to harvest fish while directed angling by fisheries managers, often partnered with anglers, exerts a known amount of angling effort.

Angling regulations are generally designed to limit fish harvest and consequently limit exploitation. Thus, relaxed harvest regulations would theoretically permit higher rates of exploitation and thus reduce the target fish abundance. However, angler preference largely mediates harvest within the confines of angling regulations (Goeman et al. 1993) and removal programs applying liberalized harvest regulations are likely to be successful only for non-native fish considered desirable by anglers (Wydoski and Wiley 1999). Angler preference may be based on food qualities of the fish, such as taste or sporting qualities, such as size.

In an application of liberalized angling regulations, Goeman et al. (1993) described efforts to shift the size structure of stunted northern pike populations in small (<200 ha) Minnesota lakes. The removal program involved both trapping and liberalized angling regulations. The authors reported fewer pike removed by anglers than by trap netting and concluded that liberalized bag limits were ineffective due to the lack of pike retention by anglers.

For undesirable fishes that are not likely to be favoured targets of anglers, incentives to increase angling effort and retention may contribute to management objectives. For example, in an effort to reduce the impacts of northern pikeminnow on native salmonids, Beamesderfer et al. (1996) successfully reduced northern pikeminnow abundance in the Columbia and Snake Rivers (Washington, Oregon and Idaho) via angling. From 1990 to 1995, directed angling with hook and line was conducted by staff members (mean effort of 12,700 hrs per yr, mean catch rate of 1.7 fish per hour) and through a reward-fisheries program. Anglers were paid $3 for each untagged pikeminnow and $50 for each tagged pikeminnow returned to check stations. Mean effort by anglers was 62,000 trips per year and 2.5 fish/trip. At the end of the six year program, a total of 117,079 northern pikeminnow had been captured by directed angling and 777,804 northern pikeminnows were caught via the reward fishery. Beamesderfer et al. (1996) reported

that the removal effort resulted in annual exploitation rates of 9 to 16%, which was within the target of 10 to 20% assumed necessary for a realized reduction of salmonid predation.

Moore et al. (2005) reported the results of a non-native rainbow trout removal project in a small stream in Great Smoky Mountains National Park, Tennessee. Using mandatory kill and report regulations over 13 days, a total of 228 anglers removed 250 rainbow trout. Based on electrofishing surveys immediately prior to the angling harvest, the authors estimated a 30% reduction in standing stock of fish in the study area. They also indicated that anglers selected for large individuals.

Another directed angling technique not often considered is the use of trot lines. Trot lines are passive sets of baited hook and line, set in a long array of many hooks tied to a single line (Hubert 1996). This method offers the advantage of minimal researcher attendance. Trotlines are not often uses for control of non-native fishes, perhaps due to highly variable catch rates (Hubert 1996).

Variability in both active and passive angling catches, as well as size-selective bias can be due in part to characteristics of the angling gears. For example, large hooks are more prone to catch large fish while small hooks are more efficient at hooking small fish (Payer et al. 1989, Ralston 1990).

Again, an understanding of the biology and ecology of target species may provide insight on the applicability of angling as a removal technique. For example, in late spring male smallmouth bass excavate a nest in shallow littoral zone water and females deposit eggs in the middle of the depression. Males guard the eggs and fry for several weeks (Scott and Crossman 1973). During this time, smallmouth bass are highly susceptible to angling (Hanson et al. 2007, Suski and Phillip 2004) and a large number of bass could be captured and removed during this period. Fisheries managers in the state of Maine (D. Boucher, Biologist, Maine Dept. of Inland Fisheries and Wildlife, pers. comm.) and in Nova Scotia have used angling to target non-native smallmouth bass as part of sampling or reclamation programs.

In summary, angling as a removal technique for non-native fish management warrants consideration. However, because angling is highly selective for large fish and volunteer removal via angling is largely dictated by angler preference, careful consideration is required to match angling removal methods to the goals of removal programs. A positive attribute of angling is that only target fish abundance is likely to be reduced provided angling gears are tailored accordingly. An example of 100% eradication resulting from angling efforts only has not been reported.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of angling for fish removal.	
Pro	***Contra***
• Public acceptance high • Public involvement • Highly selective • Benign	• Low efficiency • Required multiple efforts spanning multiple years (labour intensive)

3.3 Piscicides

The earliest reported application of piscicides to sample or control fish was in 1913 when copper sulfate was applied to a small Vermont lake to remove undesirable predatory fish (Titcomb 1914). In the early 1930's efforts to remove undesirable fish from lakes in southern Nova Scotia, again using copper sulfate, validated Titcomb's observations (Catt 1934, Smith 1938, 1940a, 1940b). Today, piscicides are commonly used to control undesirable, non-native fish (Lennon et al. 1971). Chemical reclamation is the most commonly used control/eradication technique used by fisheries managers (Wydoski and Wiley 1999, Meronek et al. 1996, Carpenter and Terrell 2005).

As many as 30 to 45 chemicals have been reportedly used or evaluated as piscicides (Dawson and Kolar 2003, Finlayson 2001). Currently, only four piscicide are approved for use in Canada and the United states, two of which are the general purpose piscicides, rotenone and antimycin A (Berger et al. 1969, Finlayson 2001, Dawson and Kolar 2003). Only rotenone and antimycin are currently widely used in North America. The two other approved piscicides are taxon-specific toxicants, Bayluside and 3-trifluoromethyl-4-nitrophenol (TFM), both of which are used for control of sea lamprey (Dawson and Koler 2003).

Rotenone has been used since 1934 (Ball 1945, American Fisheries Society 2000) when it was applied to a small pond in Michigan to remove nuisance carp. Following that initial application, thirty-two small Michigan lakes were treated with rotenone from 1934 to 1942, sixteen of which had 100% eradication (Ball 1945). Other early applications of rotenone were summarized by Smith (1940a).

Rotenone is derived from the roots of several bean-like plants (*Derris* spp.) native to South America (Ball 1945, Sousa et al 1989). Contrary to the common belief that rotenone suffocates fish, rotenone acts at the cellular level to inhibit oxidative phosphorylation, which prevents fish from using molecular oxygen for metabolic processes (Sousa et al. 1989).

Rotenone and antimycin are similar, although antimycin is an antibiotic that interrupts cellular respiration (Wydoski and Wiley 1999). The efficacy of rotenone and antimycin is similar although one important difference between rotenone and antimycin is that the latter does not elicit an avoidance response in fish, i.e. is not detectable by fish (Wydoski and Wiley 1999). In a review of 69 projects employing rotenone and 67 projects using antimycin A, Meronek et al. (1996) indicated that 48% reported success using rotenone and 45% reported success using antimycin A. Therefore, in the interest in brevity, applications of both piscicides will be discussed interchangeably.

Most applications of rotenone, as high as 97.5% of all projects, are to lentic habitats (McClay 2000, Hisata 2002), though it has been used on small streams and rivers (i.e. Baker et al. 2008, Moore et al. 2005, Carpenter and Terrell 2005). Conversely, antimycin is currently available only in a liquid form, which effectively precludes its use in deep water environments often associated with lakes and ponds (Wydoski and Wiley 1999). As such, antimycin is applied primarily to lotic systems and has been used in several stream reclamation initiatives (Carpenter and Terrell 2005, Gresswell 1991).

Moving water warrants additional precautions as piscicides if often carried outside the intended treatment area, requiring detoxification stations at the downstream extent of the treatment stream reach. The most commonly used detoxifying agent is potassium permanganate as it effectively

neutralizes the effects of rotenone and antimycin (Archer 2001, Marking and Bills, Moore et al. 2005), though the products of potassium permanganate may themselves be toxic to organisms in high concentrations or occasionally at low temperatures (Marking and Bills 1975, Lawrence 1956). Other detoxifying agents include activated carbon, chlorine and methyl blue (Hinson 2000), though the popularity of these neutralizers is low, as reflected in the literature.

Chemical reclamation projects are most often conducted on small systems (Ling 2001), primarily because of ease of treatment and cost. Reports on chemical control or eradication projects are numerous, and reported results are mixed.

For example, in a description of rotenone treatments on 33 lakes in Alberta, eradication of one or more species of undesirable fishes occurred for 60% of treatments (Alberta Sustainable Resource Development 2008). Demong (2001) described efforts to restore Adirondack strain brook trout throughout the State of New York. At least 169 restoration projects using rotenone were reported for New York State alone. Although the success rate at which undesirable fishes were eradicated was not provided, due primarily to few follow-up surveys, this study indicated that a large portion of the projects were considered successful. Similarly, Martinez (2004) reviews non-native removal efforts in ponds near the Colorado and Gunnison Rivers, Colorado. Of 86 ponds subjected to non-native fish control (several control methods), 71 had all fish eradicated, 69 of which were treated with rotenone.

There have also been large-scale chemical reclamation efforts, though they are comparatively rare and results were generally poor. For example, one of largest chemical treatments reported was the Strawberry Valley project in Utah, where in an attempt to improve angling the entire drainage basin of the Strawberry reservoir was treated with rotenone to eradicate undesirable fishes (Lentsch et al. 2001, AFS 2000). A total of 151 tributaries, with a combined treated length of 260 km, the entire epilimnion in the reservoir (volume = 3.7×10^8 m^3 and 4,872 ha surface area), and numerous spring habitats were treated to cover the entire 440 km^2 drainage basin. Lentsch et al. (2001) indicate that the two main target fishes, Utah chub and Utah sucker have reappeared but had not sufficiently colonized the area to impact fisheries. As a whole, the project was considered successful as angler catch, effort and the size of salmonid catch all increased. The total project cost was estimated at US $3.8 million.

Lake Davis, California, is a 1,619 ha (volume = 6.2×10^7 m^3) reservoir in Plumas National Forest, California. Following discovery of non-native northern pike in 1994, fisheries managers proposed a large scale eradication program using rotenone (Lee 2001). Lee reported substantial opposition to the project by the public, including restraining orders and protests; however after deliberation and the release of an environmental impact assessment, treatment began. Seventeen months following treatment, northern pike were once again discovered in Lake Davis, though it is uncertain whether they survived the eradication attempt or whether they were reintroduced.

As a whole, chemical reclamations are successful, especially when properly planned and executed and of a manageable size. For example, introduced Chain Pickerel were successfully eradicated from a small lake in New Brunswick following application of rotenone (Connell et al. 2002).

Partial control of undesirable fish species is most often achieved with piscicides, though in many cases, 100% eradication is the intended outcome of poison applications. For example, in surveys of intended objectives of piscicide applications, Wydoski and Wiley (1999) indicated that 86% of fisheries managers listed eradication as their reason for chemical treatments. Conversely,

McClay (2000) surveyed 78 states, provincial and federal agencies responsible for fisheries management and found that only 3% of treatments used piscicides for the purpose of eradication.

Inter-specific differences in fish toxicity to rotenone have been reported (Marking and Bills 1976, Meadows 1973), though in general rotenone is not considered species-specific (Dawson and Kolar 2003, Finlayson 2001). As a result, the application of piscicides will affect target and non-target fishes alike and managers should assume that all fish in the treatment area will be affected.

Aquatic organisms other than fish may be affected by rotenone and antimycin. Zooplankton communities are often affected by the application of rotenone, with cladocerans, copepods and rotifers (to a lesser degree) some of the more sensitive taxa (Brown and Ball 1943, Finlayson et al. 2000, Ling 2001). Zooplankton often recolonize treated waters (Brown and Ball 1943), though the rate of recolonization presumably depends on the proximity of a donor source and taxon-specific biology. The aquatic invertebrate community (primarily insects, molluscs and leeches) may also be heavily affected by the use of piscicides. Researchers have observed declines in invertebrate organisms following piscicide application (Arnekleiv et al. 2001, Dinger and Marks 2007, Morrison 1979 , Smith 1940a), and some taxa appear more sensitive than others (Brown and Ball 1943, Chandler and Marking 1982, Meadows 1973, Morrison 1977) which may result in community composition shifts. As with zooplankton, invertebrates may recolonize an area post-treatment, although it has been observed that certain taxa may be locally extirpated at high concentrations of piscicide (Beal and Anderson 1993, Dinger and Marks 2007).

Higher-level taxa may also be affected by piscicides but the effects are often limited to organism with highly permeable skin used for respiration (ie. amphibians) when applying rotenone (AFS 2000), Antimycin A (Turner et al. 2007), or squoxin (a selective piscicide for northern pikeminnow)(Dawson 2003). In circumstances where piscicide applications may contact amphibians that are important, endangered or of concern, potential impacts should be carefully evaluated.

The risks of rotenone or antimycin to human is considered low. Ling (2001) indicated that humans and air-breathing wildlife are "insensitive" to rotenone, and the substantial discrepancy between concentrations required to manage fish and those that affect air-breathing animals provides a larger margin of safety. No reports of fatalities have been associated with "normal" applications of rotenone, though a small child who ingested a substantial quantity of the insecticide "Galicide", which contains rotenone, did succumb to respiratory arrest (Ling 2001). In a study where rats and dogs were given only antimycin-treated water to drink and fed fish killed by antimycin (as half their diet) over a three month period, no adverse effects were documented (Gresselin and Herr 1974).

Several factors affect the toxicity of rotenone and antimycin or their treatment efficacy. Water quality parameters such as temperature, turbidity and pH have been shown to affect the rate of detoxification (Bettoli and Maceina 1996, Finlayson et al. 2000, Marking and Bills 1976). Additionally, hydrologic parameters such as stream gradient or the amount of mixing may also affect piscicide efficacy (Baker et al. 2008, Moore et al. 2008). Wydoski and Wiley (1999) report that only 35% of all chemical reclamation projects are effective for more than 10 years. Recolonization from adjacent areas is often reported as the cause of failure (i.e. Martinez 2004), and as such removal projects pairing chemical treatments with barriers to immigration may prolong the success of chemical reclamations.

In summary, many researchers and fisheries managers list rotenone as the best option for removal of undesirable, non-native fish (Demong 2001, Dawson and Kolar 2003) due to its effectiveness, ease of application, and immediate results (Lennon et al. 1971, Baker et al. 2008). With careful planning, piscicide application can be an effective non-native fish control technique. Furthermore, effects on non-target organisms may be mitigated using programs to identify at-risk organisms in the intended treatment area, collecting specimens to be held in separation (untreated areas) and repatriation programs.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of piscicides.	
Pro	***Contra***
• Applied over short time period • Can be adapted to most habitat types • Highly effective	• Impacts on other organisms • Public acceptance generally low • Costs occurred up front • Not always 100% effective

3.4 Biological Control and Eradication Techniques

Biological methods for control or eradication of fish can be loosely described as the introduction of organisms, the introduction of altered organisms or the physiological alteration of target fish in an effort to increase mortality, decrease recruitment or generally reduce the abundance of the target undesired fish. The most common and tested method is the introduction of a natural enemy. Sterilization of undesirable fish has also been attempted to hinder reproduction, reduce recruitment and lower abundance. Emerging genetic techniques to control undesirable pest species also show promise, though their use is limited and under-explored.

3.4.1 Introduction of predatory organisms

By far, the most common and promising biological control effort applied in the management of pest species is the introduction of predatory organisms (Wydoski and Wiley 1999). Several examples of introduced predators successfully controlling pest species have been documented for terrestrial systems (*see* Hoddle 2004), however examples in the aquatic world are limited.

One of the earliest records chronicling the introduction of a predatory organism for the control of undesirable fish occurred in Merced Lake, California. Smith 1896 (*in* Cole 1904) described the introduction of 19 seals (presumed by Cole to be sea lions) to the shallow lake in an effort to eradicate non-native common carp. The sea lions proved to be successful (when combined with netting efforts) and during follow up surveys, no carp were found in the lake.

More commonly, the introduced predators are fish. As such, the introduction of predatory fishes is usually intended to control small, non-native, non-game fishes that are vulnerable to predation. For example, Rumsey et al. (2007) described a 30 ha Ontario Lake where the native Brook Trout population had been severely diminished as a result of overexploitation, and yellow perch overpopulated and then stunted. In an effort to reduce the density of small perch, first generation Splake hybrids (Brook Trout x Lake Trout) were introduced. Splake effectively reduced yellow

perch abundance by 95%, although the brook trout population did not respond as expected. This study also described the elimination of rainbow smelt in Thissell Pond, Maine as a result of Splake introduction. Similarly, repeated introductions of Northern Pike into a 344 ha lake led to the collapse of a Yellow Perch population (Anderson and Schupp 1986).

Several important considerations are necessary prior to releasing a predatory species for biological control of non-native fish including; the ecological requirements of the intended predator, the vulnerability of the targeted fish to predation and the potential for development of a self-sustaining population of predatory fish (Wydoski and Wiley 1999).

The introduction of non-native fishes for the purpose of controlling other non-native fish is considered "high-risk" by many fisheries managers, primarily due to potential inadvertent impacts on non-target organisms (Hoddle 2004). Furthermore, interactions between introduced predatory fish and existing native and non-native fishes may be difficult to predict. For generalist predators, resources other than the target species may be used or even favoured (Hoddle 2004).

The introduction of predatory fish may be a viable option for the control of non-native, small-bodied, non-game fish, however the ability to control larger non-native fish appears limited, and 100% eradication is not likely. For this reason, and considering the risk, the introduction of predatory organism is not a preferred option.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of the introduction of predatory organisms as a removal technique.	
Pro	***Contra***
• Inexpensive • Can be effective at controlling target fish population • May offer additional recreational angling opportunities	• Difficult to control released organisms unless sterile • Generally unable to eradicate target fish • Species and habitat specific • Potentially harm non-target fishes • Difficult to establish and maintain predator populations

3.4.2 Other biological control techniques

Other control or removal techniques have been examined or proposed, although several technological hurdles have prevented their widespread evaluation. Of the other biological control options, only sterilization has been adequately examined. By sterilizing either males or females of a fish species, and releasing them to spawn with fertile individuals of the opposite sex, population-wide recruitment may be reduced as viable gametes from fertile individuals are "wasted" on sterile individuals.

One of the few examples deploying and evaluating sterilization is the sea lamprey control program on the St. Mary's River (Ontario-Michigan) (Great Lakes Fishery Commission 2009). The program captures lamprey in traps, retains females and releases males that have been sterilized using an automated sterilization technique (Bergstedt and Twohey 2007). From 1991 to 1996, an average of 4 600 males were sterilized and released back into the river, and from 1997

to 2004, the mean number increased to 26,500 (Bergstedt and Twohey 2007). The authors reported that reductions in theoretical reproduction averaged 58% from 1991 to 1996 and averaged 86% from 1997 to 2004.

Sterilized male release will be successful only if males can be sterilized (techniques have yet to be developed for many fish), and compensatory growth or survival does not offset recruitment reductions (Knipling 1965, Bergstedt and Twohey 2007). Because sterilization techniques are under-developed for other fish species, sterilization programs for other non-native fish have not been described.

Other methods of reducing reproductive success have been suggested. For example, Hinds and Pech (1997) describe the potential use of an immuno-contraceptive paired with viruses engineered with recombinant genes for control of non-native rabbits in Australia. They describe the technique as;

> *"the approach involves delivery of a gene expressing an essential species-specific reproductive protein in a species-specific virus, such that, when the host is infected by the recombinant virus, its immune system raises antibodies to the same proteins in its reproductive tract, blocking reproduction."* (Hinds and Perch 1997)

Hinds and Pech (1997) also explored the potential for using a similar approach to control non-native common carp, although they concluded that considerable research was required prior to contraception or contraception plus viral recombinant genes can be developed and applied.

Genetic manipulation of undesirable target organisms or the genetic engineering of viruses and infectious disease has been proposed for pest management (Thresher 2007, Thresher 2008, Kapuscinski and Patronski 2005, Crane and Eaton 1999). The introduction of pathogens for fish control is considered risky and is not a widely used control technique (Wydoski and Wiley 1999). However, several authors have suggested that pathogens, such as viruses and infectious diseases, may potentially be used to control undesirable, non-native fish. For example, Crane and Eaton (1997) explored the potential to control non-native common carp in Australia by introducing Spring Viraemia of Carp Virus (SVCV) (*Rhabdovirus carpio*), a known hemorrhagic disease in farmed carp. After thoroughly considering several factors that could influence the utility of SVCV, a significant amount of uncertainty remained. The potential to genetically engineer SVCV has recently been made possible due to technological advances and an engineered virus may provide more desirable results (Crane and Eaton 1997).

Some authors have suggested that pathogens, such as viruses and infectious diseases, may be genetically engineered to provide species specific, effective pest control options, though again, many technological advances are required prior to their application and currently, they are considered risky and / or ineffective (Muir and Howard 2004, Thresher 2008).

The use of recombinant genes has also been suggested for pest management, and in particular, autocidal genes have generated interest as they can be species-specific, target specific life stages, and are potentially reversible (Thresher 2008). Autocidal genes can be modified such that managers can control its impacts or abundance (Gould and Schliekelman 2004). Autocidal genes, as summarized by Thresher (2008), may be used for stage- or sex-specific sterility or lethality,

gender distortion, inducible mortality, pleiotropy (aka. Trojan genes) and selfish genes. As with the other biological options, recombinant genes are untested.

Fisheries and wildlife managers faced with the dubious task of controlling or eradicating undesirable, non-native species are bound by current and accepted methodologies. These developing biological control methods are not yet viable options. However, based on theoretical work and rapidly expanding technologies, these proposed control methods may in the near future offer a distinct advantage over other control measures.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of other biological fish removal techniques.	
Pro	***Contra***
• Genetic processes potentially inexpensive • Potentially species-specific	• Largely untested • Potentially difficult to control pathogens once introduced • Possible transfer of pathogens to non-target species • Legal issues • Unknown public acceptance • Sterilization expensive

3.5 Environmental Manipulation

Environmental control of fish involves the manipulation of habitat or a fish's environment to evoke a negative response that ultimately reduces population abundance of the target species. Currently, only water level manipulations and the creation of "winter-kill" conditions have been explored as possible environmental techniques to control undesirable fishes.

The most widely used environmental manipulation for the control of undesirable fishes is complete dewatering of aquatic habitat. Dewatering is the best method to ensure 100% fish eradication, and the efficacy of other fish removal options are often compared to dewatering (American Fisheries Society 2000, Finlayson et al. 2002, Ling 2001, McClay 2000). However, in many natural habitats (i.e. non-regulated), the option to drain the system is impractical, particularly for lotic systems and natural lakes. Furthermore, effective disposal or treatment of drained water is necessary to prohibit non-native fish from spreading to adjacent systems, and this may be logistically challenging (Maryland Secretary of Natural Resources 2002).

Partial dewatering of lentic habitats can be considered water level manipulation and may inhibit reproductive success (Neves 1975) or benefit other control options. For example, largemouth bass movements and habits were altered following water level manipulations (Rogers and Bergersen 1995), which could potentially alter their vulnerability to other removal techniques (i.e. piscicide applications, electrofishing). Also, lowered water level has been shown to increase feeding in some predators (Heman et al. 1969), and may increase susceptibility to predation for prey fish or juvenile undesirable fishes.

In lotic habitats, the manipulation of flow has been suggested as an effective option to control reproduction in some fishes. Based on observations regarding smallmouth bass nesting and habitat in the Rapid River, Maine, Boucher (2005) suggested that flow manipulation should be examined as a tool to decrease bass recruitment and growth. In 2007, experiments to assess the impacts of pulsed water releases on non-native smallmouth bass nesting success in the Rapid River indicated that this approach has potential as a control measure (Kleinschmidt 2007).

Winterkill is a phenomenon that occurs when oxygen is reduced due to extended periods of ice and snow cover and when depth is insufficient to provide a large volume of water (Lennon 1971, Summerfelt 1999). Induced winterkills have been used to control undesirable fishes with fairly good success (Alberta Sustainable Resource Development 2008, Verrill and Berry 1995, Shroyer 2007). Winterkills as a management tool appear to be restricted to shallow lakes of moderate to high fertility.

Environmental methods of fish control or eradication are effective under certain conditions and where applicable, should be considered for any removal project.

Summary of positive attributes (*Pro*) and limitations (*Contra*) of environmental removal techniques.	
Pro	***Contra***
• Highly effective	• Not feasible in all situations
• May be low cost	• May affect non-target species
• May require little labour	• Environmentally disruptive
• Public acceptance mod. to high	• Logistics of water disposal

3.6 Summary of Control and Eradication

As discussed in section 3.1, the selection of appropriate control or eradication techniques must address the potential for target fish to be affected and determine whether the biological benefits (i.e. reduced predation on native fishes, elimination of the potential for spread) outweigh the financial costs and social or political implications (Beamesderfer 2000, Chadderton 2003). In addition to biological benefits, the selection of control technique will largely depend on its efficacy giver the biology/ ecology of target fish, the type of habitat being managed, and potential side-effects on native or non-target biota.

Removal techniques may in theory provide the proper set of criteria and may be well suited for specific objectives, however there are few reported successes with this technique. In some cases, a failure to remove the target undesired fish may have dire consequences (for example: when protecting endangered native fish), and as such the risk associated with using unproven removal techniques may be unacceptable. Although fish control and eradication projects have a century of history, the diversity of removal methods is vastly skewed towards only a few methods, many new and promising techniques remain untested.

Based on the preceding literature review, the flowing generalizations can be made regarding the major divisions of removal techniques.

Mechanical removal techniques are effective control options that use readily available equipment however they are often labour intensive (Finlayson et al. 2000, Good and Cargnelli 2004, Maine Department of Inland Fisheries and Wildlife 2006) and mostly effective on fish of low reproductive potential (Parker et al. 2001, Knapp and Mathews 1998, Vredenberg 2004). For the objective of eradication, mechanical removal has only been demonstrated in small, closed systems (Kulp and Moore 2005, Baker et al. 2008).

Chemical removal techniques (piscicides) can be highly effective and require relatively little labour. However, substantial side effects on non-target fish and invertebrates have been reported (Brown and Ball 1943, Chandler and Marking 1982) and public acceptance of piscicides is often low (Finlayson et al. 2000, McClay 2000).

Biological removal techniques are largely untested and under-developed. Several techniques have been proposed, though only male sterilization has been adequately tested in-field. Biological control techniques are often considered "risky" (Finlayson et al. 2000, Maine Department of Inland Fisheries and Wildlife 2006, Kapuscinski and Patronski 2005), though several authors have suggested their potential benefits (Roberts and Tilzey 1997, Thresher 2008, Kapuscinski and Patronski 2005).

Environmental removal techniques have been successfully used to control and eradicate fish, though the latter has been achieved primarily via complete dewatering. The adverse effects on non-target aquatic organisms may be substantial, though public acceptance may be high (Finalyson et al. 2000). Flow manipulation in lotic environments is promising yet remains to be adequately tested (*see* Kleinschmidt Energy and Water Resource Consultants 2008).

Several reviews of fish eradication or control projects have been compiled. Most have focused primarily on chemical piscicides. In one of the most comprehensive, multi-technique reviews Meronek et al. (1996) reviewed 250 fish control or eradication projects, indicating that 63% of the projects used chemical reclamation methods, 30% of the projects implemented physical removal or water level drawdown, and the remainder used some form of fish stocking or any combination of the aforementioned. This prevalence and preference for chemical piscicides is further illustrated as other reviews of fish management projects focused primarily on piscicides (Dawson and Kolar 2003) or solely on piscicides (Lennon et al 1971).

Dawson and Kolar (2003) also reviewed several removal techniques, though only briefly, and piscicides were the primary focus. They indicated that general chemical piscicides were the best option to control undesirable fishes, particularly in situations where removal is critical to the persistence of native species. In situations where the removal of undesirable fishes is "less critical", they suggested that integrated removal techniques, using several technologies, may be better suited.

Similarly, earlier work by Lennon et al. (1971) focused on piscicides and reviewed the chemical reclamation projects. Their survey of 159 fisheries agencies from around the world revealed that the management of undesirable fishes using chemical piscicides was widespread and effective. They also concluded that the use of chemical piscicides should be restricted to situations where practical alternatives (i.e. other removal methods) are unsuitable. Also, they suggest that the use of barriers in conjunction with removal projects may prevent re-invasion and thus extend the effectiveness of reclamation actions.

Multiple integrated approaches are commonly suggested by several authors, fisheries managers, and biologists to increase the likelihood of successful control or eradication of undesirable fish (Dawson and Kolar 2003, Meronek et al. 1996). However, with the exception of pairing barriers with removal, few multiple control / eradication techniques have been evaluated.

Arguably the largest example of a successful large scale control program deploying multiple techniques is the ongoing sea lamprey control program, administered by the Great Lakes Fishery Commission. They deploy the lampricides "Bayluscide", operate in excess of 50 lowhead, velocity or electric barriers, use trap nets to capture sexually-mature adult lampreys and release sterile males for biological control (Great Lakes Fishery Commission 2009, Holmes 1995). This extensive program, covering all five Great Lakes, has resulted in an estimated 90% reduction in the abundance of sea lamprey (Great Lakes Fishery Commission 2009).

Finally, authors commonly suggested that sufficient commitment and intensity must be administered (Tyus and Saunders 2000, Carpenter and Terrell 2005, Bomford and Tilzey 1997). Several control programs were successful in immediate reductions in abundance, but failed to achieve a long-term reduction because of a lack of commitment or effort (Earle and Lajeunesse 2007, Burdick 2008, Zipkin et al. 2008).

In conclusion, if control of undesirable fish is sufficient to achieve management goals then several removal options are available. Several examples of most technologies are reported and a summary of their applicability is presented in Table 2. The intensity and duration of their application can be manipulated to deliver varying rates of removal and address habitat or politically-driven restrictions.

If eradication is the chosen management objective, several researchers conclude that aside from complete dewatering, chemical (rotenone or antimycin) treatment offers the highest probability to achieve complete eradication of undesirable fishes (Finlayson et al. 2002, Ling 2001, McClay 2000, Maine Department of Inland Fisheries and Wildlife 2005). Conversations with numerous fisheries biologist and fisheries managers, confirm this assertion, indicating that this opinion remains current and widespread.

4.0 MIRAMICHI LAKE SMALLMOUTH BASS MANAGEMENT OPTIONS

4.1 Description of Miramichi Lake

Miramichi Lake (46.46° N, 66.97° W) is situated in the upper drainage basin of the Southwest Miramichi River, New Brunswick (Fig. 1). It measures 221 hectares in surface area, 2.8 km in maximum axial length, approximately 0.8 km wide and has approximately 8.6 km of shoreline. Miramichi Lake has a mean depth of 2.4 m while the maximum depth is 7.3 m and records indicate that the lake does not stratify (*see* Cronin 2008). The lake is slightly acidic with summer pH between 5.4 and 6.0. The drainage basin measures 46.5 km^2 with inflows to the lake comprising of two first order streams and a second order stream (at 1:50,000 scale). Miramichi Lake is connected to the Southwest Miramichi River via Lake Brook, which is 5.3 km in length, has a mean width of approx. 8 m, an estimated mean annual flow (MAF) of approx. 1.06 $m^3 \cdot sec^{-1}$ and a maximum estimated flow (2 year flood) of approx. 11.4 $m^3 \cdot sec^{-1}$ (D. Cassie, DFO, *unpublished data,* Cronin 2008).

Known fish species in the lake include; anadromous Atlantic salmon, brook trout, white perch; white sucker, lake chub, creek chub yellow perch, fallfish, common shiner, golden shiner, and now smallmouth bass (Cronin 2008).

4.2 Options for Managing Smallmouth Bass in Miramichi Lake

Reports of interactions of smallmouth bass and native fishes in other parts of North America have indicated that smallmouth bass can severely alter food web structure (Lepak et al. 2006, Vander Zanden et al. 2004), reduce fish biodiversity (Jackson 2002) and contribute to increased mortality of native fishes (Weidel et al. 2007, Fayram and Sibley 2000, Katano and Aonuma 2002). The concern is the potential for smallmouth bass to spread from Miramichi Lake and to colonize the Miramichi River system, an event which may have already occurred, although unverified.

Managing introduced smallmouth bass is a difficult and substantial undertaking, and efforts in other jurisdictions have reported mixed results (Weidel et al. 2007, Weedman and Sponholtz, Moyle et al. 1983). The early detection of smallmouth bass in Miramichi Lake is a significant advantage, and immediate, direct and intensive approaches will likely offer the greatest potential for mitigation. An integrated, multi-technique approach to the management of non-native fishes is this author's recommended strategy to achieve containment, control or eradication objectives (Dawson and Kolar 2003, Bergstedt and Twohey 2007, Popper and Carlson 1998). A management program on this system should encompass four operational phases; precautionary containment, a detailed assessment of distribution, control or eradication efforts and post-treatment monitoring.

4.2.1 Phase 1 – precautionary containment

The opportunity for effective control of undesirable species decreases as populations establish, expand and become established (Owen 1998 and Chadderton 2003) (Figure 2). Thus, every

effort should be made to ensure containment of smallmouth within their current known distribution.

Electrofishing surveys have indicated that smallmouth have not yet colonized Lake Brook below the initial 300m (slow, deep, lake-like habitat) (D. Moore, DFO, *unpublished data*) and as such this is the most obvious location for a barrier.

In the context of reducing entrainment at hydroelectric plants, Turnpenny et al. (1998) indicate that physical screens offer the highest guaranteed fish diversion efficiencies (when compared to behavioural barriers) and may be the most cost effective barrier option for small turbine intakes of less than 1 $m^3 \cdot sec^{-1}$. This could be considered to apply to barriers for downstream migrating fishes in natural environments as well. Additionally, the mean annual outflow of 1.06 $m^3 \cdot sec^{-1}$ falls closely to the recommended range, although spring flows would be greater than this.

Physical barriers are considered best suited for the outflow location. A physical barrier can be effective, inexpensive on this small scale and due to the remoteness of the site, behavioural barriers requiring electricity are not applicable. A screen may prevent movement through the slow sections of the river, however considerable debris would be expected to accumulate on the screen and daily maintenance would be required. Screens placed at the transition from deep lotic habitat to fast lentic habitat may facilitate debris removal and also allow the opportunity to trap migrant fish.

Serious consideration should also be given to the installation of two barriers, separated by some distance. The application of two barriers offers several advantages. First, should one barrier fail, the second may retain migrating fish. Fish can them be removed from the stream section between the two barriers. Secondly, by locating a barrier at the lake's outlet and another further down in the system, smallmouth bass that have not been detected due to their low densities and occupy the stream immediately below the lake may be contained. Several authors have described the benefits of multiple barriers and prescribed their application (Carpenter and Terrell 2005).

Barriers should also be considered on the three small tributaries that feed into the lake. Juvenile bass may use these areas to feed and large bass may find refuge in these brooks if chemical piscicides (rotenone in particular) are applied to the lake. If eradication is to be attempted in the lake, the re-colonization of the lake from donor sources such as these brooks may jeopardize the program. These small tributaries appear well suited for hydraulic barriers and in areas of local high gradient, small temporary artificial waterfalls may inhibit the upstream movement of bass. Again, paired barriers would be recommended.

Provisions to accommodate the movements of native salmonid fishes, in particular seaward salmon smolts, salmon kelts and anadromous brook trout, should be associated with the downstream barrier, and to some extent the barriers in the small tributaries. The use of fyke nets during peak migration periods may permit these fish to be captured prior to reaching the barriers. Captured salmonids could then be placed downstream of the barrier to continue on their migration. Captured non-native species could be destroyed. Similarly, fyke nets could be used immediately downstream of the barrier to capture native fishes migrating upstream.

Barriers should be constructed prior to warming of the water as increasing temperature may increase bass movements and the likelihood of dispersal.

4.2.2 Phase 2 – detailed assessment of distribution

A thorough understanding of the distribution of smallmouth bass in Miramichi Lake and the surrounding streams is essential to guide management decisions. Based on electrofishing surveys in October 2008, smallmouth bass have been captured only in the lentic habitats of Miramichi Lake (D. Moore, DFO, *unpublished data*). However, following the installation of barriers, additional comprehensive surveys should be conducted to reaffirm the absence of smallmouth bass downstream from the lake. If the distribution of smallmouth is greater than anticipated, a management program may be tailored to encompass all occupied habitats.

The presence of smallmouth bass in the main Southwest Miramichi would severely diminish the likelihood of a successful eradication or control program as the large size of the main river precludes many control methods.

4.2.3 Phase 3 – control or eradication

Management actions that reduce the abundance of smallmouth bass in Miramichi Lake would likely reduce the negative impact on native organisms. The most promising removal options include:

- Repeated, annual boat electrofishing
- Repeated, annual trapping and netting efforts
- Directed angling targeting spawning smallmouth bass
- A reward-fishery for smallmouth bass
- Liberalized angling regulations
- Water level manipulation during the nesting period

Annual boat electrofishing or trapping efforts may be successful in collecting a large number of smallmouth bass, though trapping is likely to be only marginally effective as efforts for smallmouth bass are generally unsuccessful (Boucher 2008, Wright 2000). Furthermore, a perpetual removal effort is likely to be costly and labour intensive.

Angling may also be able to remove a large number of smallmouth bass, particularly adult male bass as they are highly vulnerable during the nesting period of May and June (Hanson et al. 2007, Lukas and Orth 1995). The removal of male bass during the spawning season may reduce nesting success and the recruitment of fry to the population. Directed angling is the most promising option as volunteer angler effort is unpredictable, and the retention rate of smallmouth bass is generally low (Nova Scotia Department of Fisheries and Aquaculture 2001).

For several reasons, the option of control, as opposed to eradication, is less likely to result in the elimination of potential negative impacts of the smallmouth bass introduction.

First, compensatory response to removal of smallmouth bass has been demonstrated. An intensive, seven year electrofishing removal effort on a 271 ha lake in New York was successful in removing 54,000 smallmouth bass between 2000 to 2007 (Weidel et al. 2007). The biomass of smallmouth bass was reduced, however their abundance increased with most of the increase comprised of young-of-year, yearling and sub-adult bass (Zipkin et al. 2008). Zipkin et al. (2008) suggested that a compensatory response to removal, and a release from predation are the likely

mechanisms for this increase despite intensive removal efforts. Also, the ability to detect, capture and remove smallmouth bass at low densities may be low and complete eradication would not likely occur.

Finally, even if control is achieved and abundance is reduced, the likelihood of smallmouth bass emigrating from the lake and colonizing additional areas within the watershed increases with time, even with the construction of barriers. Several authors have suggested that containment and control programs only delay the effects of non-native fishes or their dispersal to new habitats (Carpenter and Terrell 2005).

For the above reasons, partial control resulting in the removal of a portion of the smallmouth bass population is not likely to eliminate the risk of impacts of smallmouth bass to Miramichi Lake nor are they likely to prevent smallmouth bass from spreading throughout the system. Complete eradication is the only option that will prevent further dispersal. As was concluded in chapter 3, chemical reclamation offers the highest potential for eradicating undesirable fish in most habitats.

In the last half century, only two eradication projects have been attempted in the Maritime Provinces, both of which were in New Brunswick and both of which used chemical reclamation as the main removal technology. The first eradication attempt was an effort to remove introduced goldfish from Killarney Lake, a small 10 ha lake located in Fredericton city limits. Hooper and Gilbert (1978) reported 2 applications of antimycin A on consecutive days in May 1974. Their efforts yielded 419 goldfish and in excess of 5,000 native fishes, and piscicide treatment effectively eradicated goldfish from the lake. They note that reinvasion of eels and cyprinid species through the lakes outlet eventually reduced the reclaimed lakes trout production potential to pre-treatment levels.

Connell et al. (2002) reported on an action to eradicate chain pickerel from Despres Lake, which drains into the Cains River, tributary of the Southwest Miramichi River. The authors reported substantial concern for the potential of pickerel to spread throughout the Miramichi system. As such, rotenone was applied to the 16.5 ha lake in October 2001 and a total of 691 chain pickerel and in excess of 3,200 yellow perch were collected. Initial post-treatment electrofishing surveys indicated that pickerel had been successfully eradicated from the system.

These experiences and those reported from other jurisdictions will be crucial in the implementation of any management plan for Miramichi Lake, though removal efforts are site- and species-specific. The moderately large size of Miramichi Lake warrants consideration for combining several approaches, including chemical reclamation and alternate approaches in other habitats. For example, in addition to chemical reclamation, electrofishing in the small tributaries which feed into Miramichi Lake will be required to assess the distribution of bass. To ensure bass are not present in these streams, intensive re-surveying should be conducted and if found, a multi-pass removal effort should be made following the examples of Kulp and Moore (2000) and Baker et al. (2008).

Additionally, in an effort to minimize the impacts of lake reclamation on non-target fishes, and amphibians, a native fish/amphibian salvaging program should run concurrently with non-native removals. In other jurisdictions, non-native fish have been successfully captured, stored in holding facilities and repatriated once conditions were conducive to their survival.

Miramichi Lake is similar in size, chemical composition, limnology and bathymetry to many other lakes where chemical reclamation has been successful. Fisheries professionals conducting the reclamation should be familiar with standard operating procedures for rotenone or antimycin application. Finlayson et al. (2000) provide a thorough review of the administrative and technical aspects of piscicide application.

4.2.4 Phase 4 - post-treatment monitoring

An often neglected phase in the management of non-native fish is post-treatment monitoring. Not only are post-treatment surveys important to assess whether additional efforts are required, but reporting of successful reclamations is important for future non-native fish management programs. Smallmouth bass in low densities may be difficult to detect, however electrofishing efforts concentrated on preferred littoral habitat provides the highest probability of detecting bass if they are present.

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TABLES

Table 1. Summary of barrier traits and suitability.

Barrier type	100% containment demonstrated?	Maximum reported effectiveness	Suitability for upstream movement	Suitability for downstream movement	Suitability for lentic habitats	Reliability	Level of maintenance required	Relative initial cost
Physical barriers								
Water Fall	Yes	100%	High	Low	Low	High	Low	Low - High
Velocity	Yes	100%	High	Low	Low	High	Low	Mod - High
Screen/nets	Yes	100%	High	High	High	High	High	Mod - High
Behavioural								
Electric	No	99%	High	Mod	High	Mod - High	Low	Mod - High
Bubble barrier	No	98%	Mod	High	High	High	Low	Low
Acoustic	No	95%	High	High	High	High	Low	Low
Light-based	No	95%	High	High	High	High	Low	Low

Table 2. Summary of control and/or eradication method traits and suitability.

Category	Method	Proven for control	Proven 100% effective for eradication	Suitability in lotic environments	Suitability in lentic environments	Level of effort required	Typical number of applications	Relative short-term cost	Risk Level
Mechanical	Electrofishing	Yes	Yes	Mod - High	High	High	Multiple	Mod	Low
Mechanical	Nets, traps and weirs	Yes	Yes	High	High	High	Multiple	Low-mod	Low
Mechanical	Explosives	No	No	Mod	High	Low	Multiple	Mod	Mod
Mechanical	Angling	Yes	No	High	High	Low – Mod	Multiple	Low	Low
Chemical	Piscicide	Yes	Yes	Mod - High	High	Low	Single	High	Mod
Biological	Species introduction	Yes	Yes	High	High	Low	Multiple	Low	High
Biological	Other Biological Methods	Yes	No	High	High	Low	Multiple	Low	High
Environ-mental	Water-level manipulation	Yes	Yes	Low[1] Mod[2]	Low[3] – High[4]	Low	One[1] – Multiple[2]	Low	Low

1 = Dewatering, 2= Water flow manipulation, 3= Natural, 4= Impoundments.

FIGURES

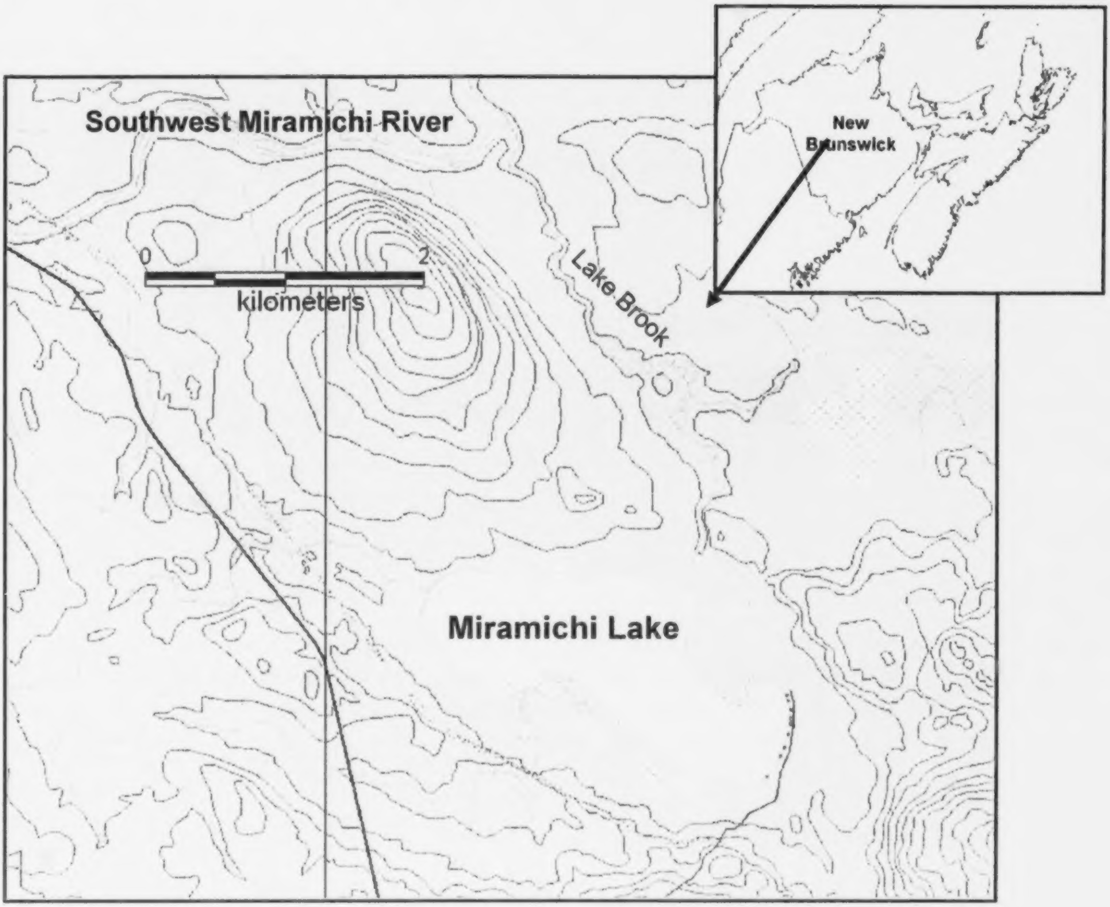

Figure 1. Map of Miramichi Lake and the Southwest Miramichi River.

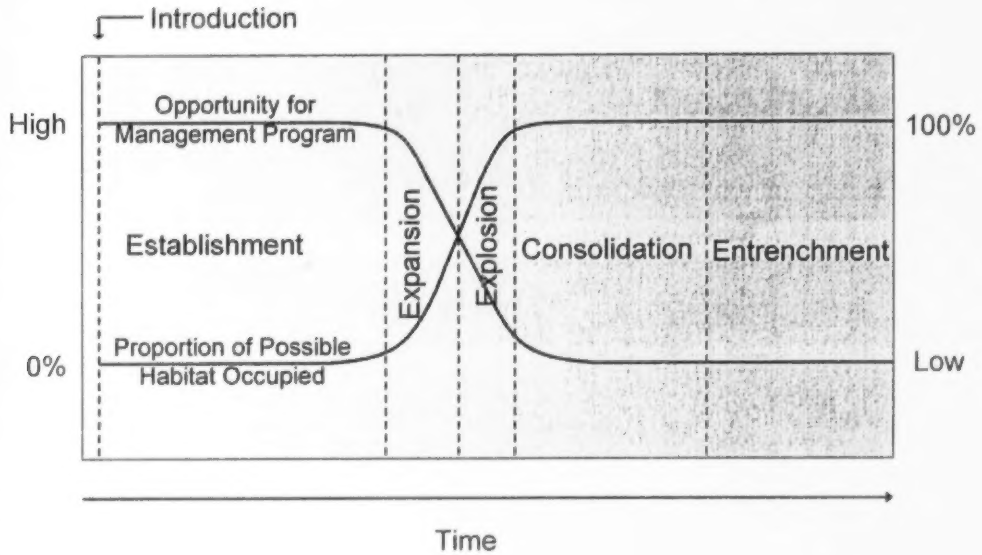

Figure 2. Relationship between the spread or establishment of non-native fish and the opportunity for a mitigative management program. Adapted from Owen (1998) and Chadderton (2003).

APPENDIX

Appendix 1. List of common names and scientific names for species referenced in this review.

Common Name	Scientific Name
Alewife	*Alosa pseudoharengus*
American Eel	*Anguilla rostrata*
Asian Carp	*See* Bighead, Grass and Silver Carps
Atlantic Cod	*Gadus morhua*
Atlantic Salmon	*Salmo salar*
Bighead Carp	*Hypophthalmichthys nobilis*
Bigmouth Buffalo	*Ictiobus cyprinellus*
Blue Tilapia	*Oreochromis aureus*
Blueback Herring	*Alosa aestivalis*
Bonneville Cutthroat Trout	*Oncorhynchus clarki utah*
Brook Trout	*Salvelinus fontinalis*
Brown Trout	*Salmo trutta*
Chinook Salmon	*Oncorhynchus tshawytscha*
Colorado River Cutthroat Trout	*Oncorhynchus clarki pleuriticus*
Common Carp	*Cyprinus carpio*
Common Shiner	*Luxilus cornutus*
Creek Chub	*Semotilus atromaculatus*
Eurasian Ruffe	*Gymnocephalus cernuus*
European Minnow	*Phoxinus phoxinus*
Fallfish	*Semotilus corporalis*
Gizzard Shad	*Dorosoma cepedianum*
Golden Shiner	*Notemigonus crysoleucas*
Goldfish	*Cyprinus auratus*
Grass Carp	*Ctenopharyngodon idella*
Green Sunfish	*Lepomis cyanellus*
Greenback Cutthroat Trout	*Oncorhynchus clarki stomias*
Kokanee Salmon	*Oncorhynchus nerka*
Lake Chub	*Couesius plumbeus*
Lake Trout	*Salvelinus namycush*

Appendix 1 (continued).

Common Name	Scientific Name
Menhaden	*Brevoortia tyrannus*
Mosquitofish	*Gambusia affinis*
Northern Pikeminnow (aka. N. Squawfish)	*Ptychocheilus oregonensis*
Rainbow Smelt	*Osmerus mordax*
Rainbow Trout	*Oncorhynchus mykiss*
Red Shiner	*Cyprinella lutrensis*
Round Goby	*Neogobius melanostomus*
Rudd	*Scardinius erythrophthalmus*
Sea Lamprey	*Petromyzon marinus*
Silver Carp	*Hypophthalmichthys molitrix*
Smallmouth Bass	*Micropterus dolomieu*
Spot	*Leiostomus xanthurus*
Threadfin Shad	*Dorosoma petenense*
Topmouth Gudgeon	*Pseudorasbora parva*
Utah Chub	*Gila atraria*
Utah Sucker	*Catostomus ardens*
Walleye	*Sander vitreus*
White Perch	*Morone americana*
White Sucker	*Catostomus commersoni*
Yellow Bullhead	*Ameiurus natalis*
Yellow Perch	*Perca flavescens*
Yellowstone Cutthroat Trout	*Oncorhynchus clarki bouvieri*